THE UNIVERSITY OF QUEENSLAND

AUSTRALIA

The Effectiveness of Rehabilitation Post Catastrophic Environmental Events as A Conservation Strategy for Marine Turtles.

Jaylene Bevlyn Flint

BSc, GradDip Ed

A thesis submitted for the degree of Doctor of Philosophy at

The University of Queensland in 2017

School of Veterinary Science

Abstract

In-water monitoring of the health and well-being of marine vertebrates is usually expensive and therefore may not be undertaken by management agencies with financial constraints. However, the use of stranding data can provide a cost-effective alternative estimation of disease and mortality. Strandings for marine turtles in Queensland are recorded in a web based database (StrandNet) managed by the Queensland Government's Department of Environment and Heritage Protection (EHP). Data recorded in StrandNet for marine turtles stranded from the entire east coast of Queensland between 1996 and 2013 were investigated for patterns of stranding in an attempt to identify which factors, such as extreme weather events, may cause stranding of marine turtle species and, further, use these patterns to predict stranding and the required responses to mitigate the negative impact mass mortalities have on endangered species such as marine turtles.

Significant stranding trends in Queensland between 1996-2013 were: (i) an increase in the number of animals reported stranded within the study site; (ii) a species (loggerhead and green marine turtles) prevalence for stranding; (iii) a seasonal effect on different age classes stranding with most overall strandings occurring between August and November; and (iv) stranding hotspots (Moreton Bay, Hervey Bay, Rockhampton region and Cleveland Bay) persisting throughout the study timeframe.

One strategy to mitigate the negative effects of marine turtle stranding is to provide medical care to those that strand alive in the hopes they can return to the functional population. Rehabilitation of marine turtles in Queensland is multifaceted. It treats individual animals, serves to educate the public, and contributes to conservation. Of 13854 marine turtles reported as stranded during the 18-year period of investigation, 5022 of these turtles stranded alive with the remainder verified as dead or of unknown condition. A total of 2970 (59%) of these live strandings were transported to a rehabilitation facility.

The original cause of stranding has an impact on the success of rehabilitation and this may influence where treatment efforts are directed. For example, of the turtles admitted to rehabilitation exhibiting signs of disease (natural cause of illness) (18% of all animals admitted to rehabilitation), 88% of them died either unassisted or by euthanasia. Sixty-six percent of turtles admitted for unknown causes of stranding died either unassisted or by

euthanasia. By contrast, all turtles recorded as having a buoyancy disorder with no other presenting problem or disorder recorded, were released alive.

One hundred and one of the turtles released from rehabilitation were reencountered: 77 reported as restrandings (20 dead, 13 alive subsequently died, 11 alive subsequently euthanized, 33 alive) and 24 recaptured during normal marine turtle population monitoring or fishing activities. Considering the high mortality rate and low successful recapture rate, rehabilitation may not be economically viable in its present configuration.

Not admitting marine turtles to rehabilitation centres and returning alive animals to sea after basic in-field triage may not address the presenting problem either. During this 18-year retrospective investigation, 1261 turtles were released back into the ocean without being admitted to rehabilitation. Of these, 67% of animals re-stranded for a second time with the same initially recorded reason.

Being able to understand commonalities of marine turtle strandings is important for marine resources managers to permit better decision-making and allocation of resources following increased strandings. Several environmental factors influence the prevalence of marine turtle stranding. These factors are thought to be rainfall, freshwater discharge and temperature. There have been links established between seagrass die off and flooding events making these chosen factors good proxies of seagrass availability/viability. Increased rainfall leads to increased freshwater discharge into the marine environment bringing with it increased nutrient and sediment loads that smother sea-grasses and other food items, directly impacting marine turtles by removing their available food sources. Similarly, for multiple underlying reasons, more strandings occur during the warmer months. Using these foundations, we can predict when and how many strandings are likely to occur by the manipulation of environmental variables in a predictive model.

Given the identification of stranding predisposition, hotspots, environmental triggers, the cost of individual treatment and the availability of alternative options, this study suggests that rehabilitation may not be viable to treat all stranded turtles, unless the cause and circumstances of stranding are historically treatable. Instead, efforts may be better used if mobile triage units are deployed to treat juvenile green turtles with unknown reasons for stranding in hotspots such as Moreton Bay, Hervey Bay, Rockhampton region and Townsville Region (Cleveland Bay) after a major flooding event has occurred. While this model is robust based on the preliminary available data, it may be adjusted to incorporate other influencing factors such as specific disease effects under catastrophic conditions and improve outcome (successful returns to the ocean) through more research into diseases and survival rates to produce more accurate predictions.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my research higher degree candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

I acknowledge that an electronic copy of my thesis must be lodged with the University Library and, subject to the policy and procedures of The University of Queensland, the thesis be made available for research and study in accordance with the Copyright Act 1968 unless a period of embargo has been approved by the Dean of the Graduate School.

I acknowledge that copyright of all material contained in my thesis resides with the copyright holder(s) of that material. Where appropriate I have obtained copyright permission from the copyright holder to reproduce material in this thesis.

Publications during candidature

Peer Reviewed Papers

Flint J, Flint M, Limpus CJ, Mills PC. Trends in Marine Turtle Strandings along the East Queensland, Australia Coast, between 1996 and 2013. J Mar Biol. 2015;2015. doi:10.1155/2015/848923

Flint M, Mills PC, Loban F, Simpson T, Lui S, Fujii R, Whap, D, Flint, J, Owen H. Development of a Humane Slaughter Device for Green Turtles for Use by Traditional Owners in the Torres Strait Islands, Australia. Fuentes MMPB, editor. PLoS One. American Veterinary Medical Association; 2017;12: e0167849. doi:10.1371/journal.pone.0167849

Owen, H., Flint, J.B., and Flint, M. (2017). Marine debris and fisheries- Sirenia. Chapter 15. Marine Mammal Welfare. Butterworth, A. (Ed). Springer Publishing, New York City. ISBN 978-3-319-46994-2

Flint J, Flint M, Limpus CJ, Mills P. Status of marine turtle rehabilitation in Queensland. PeerJ.; 2017;5: e3132. doi:10.7717/peerj.3132

Flint J, Flint M, Limpus CJ, Mills PC. The impact of environmental factors on marine turtle stranding rates. PLoS One. 2017;12: e0182548. doi:10.1371/journal.pone.0182548

Conference abstracts and presentations

Flint, J., Flint, M., Limpus, C.J., and Mills, P.C. (2015). Stranding and rehabilitation: How can it be made better? *Southeast Regional Sea Turtle Meeting*. Feb 4-7, 2015. Jekyll Island, United States.

Publications included in this thesis

Flint J, Flint M, Limpus CJ et al. Trends in marine turtle strandings along the east Queensland, Australia coast between 1996 and 2013. *J Mar Biol* 2015;7.
-incorporated as Chapter 3

Contributor	Statement of contribution
J Flint	Designed statistical analysis (90%) Edited database (95%) Wrote the paper (75%)
M Flint	Designed statistical analysis (5%) Wrote and edited paper (15%)
C Limpus	Wrote and edited paper (5%) Designed statistical analysis (5%) Conferred on database edits (5%)
P Mills	Wrote and edited paper (5%)

Flint J, Flint M, Limpus CJ, Mills P. Status of marine turtle rehabilitation in Queensland. PeerJ.; 2017;5: e3132. doi:10.7717/peerj.3132
-incorporated as Chapter 4

Contributor	Statement of contribution
J Flint	Designed statistical analysis (95%) Wrote the paper (80%)
M Flint	Wrote and edited paper (10%)
C Limpus	Wrote and edited paper (5%) Conferred on statistical analysis (5%)
P Mills	Wrote and edited paper (5%)

Flint J, Flint M, Limpus CJ et al. What happens to stranded turtles not sent to rehabilitation? Submitted to Marine Turtle Newsletter
-incorporated as Chapter 5

Contributor	Statement of contribution
J Flint	Designed statistical analysis (90%) Wrote the paper (80%)
M Flint	Designed statistical analysis (5%) Wrote and edited paper (10%)
C Limpus	Designed statistical analysis (5%) Wrote and edited paper (5%)
P Mills	Wrote and edited paper (5%)

Flint J, Flint M, Limpus CJ, Mills PC. The impact of environmental factors on marine turtle stranding rates. PLoS One. 2017;12: e0182548. doi:10.1371/journal.pone.0182548
-incorporated as Chapter 6

Contributor	Statement of contribution
J Flint	Designed statistical analysis (95%) Wrote the paper (85%)
M Flint	Designed statistical analysis (5%) Wrote and edited paper (5%)
C Limpus	Wrote and edited paper (5%)
P Mills	Wrote and edited paper (5%)

Flint J, Flint M, Limpus CJ et al. Predicting the magnitude of marine turtle strandings based on weather conditions.

-incorporated as Chapter 7

Contributor	Statement of contribution
J Flint	Designed statistical analysis (95%) Wrote the paper (85%)
M Flint	Designed statistical analysis (5%) Wrote and edited paper (5%)
C Limpus	Wrote and edited paper (5%)
P Mills	Wrote and edited paper (5%)

Contributions by others to the thesis

Dr. J Martin – Provided statistical support for chapters 6 &7. He provided on-site assistance and guidance.

Dr. M Chaloupka – was consulted about chapter 6 & 7. He provided some support and suggestions.

Statement of parts of the thesis submitted to qualify for the award of another degree

None

Acknowledgements

Firstly, I would like to thank my advisors Professor Paul Mills and Adjunct Associate Professor Col Limpus. They both assisted me by bringing skills and knowledge from their respective fields to allow me to conduct this project in the direction that I wanted, for which I am very grateful. I would also like to thank Dr. Julien Martin for his assistance and guidance through the modelling aspect of this project.

This project required assistance from many people, to all of which I am grateful. I would like to thank Underwater World -SEALIFE Aquarium (Mooloolaba), The Australia Zoo Wildlife Hospital and SeaWorld Gold Coast for allowing me access to their data and for information provided during the course of this investigation.

I would also like to acknowledge the in-kind support of The Florida Aquarium.

I am grateful for the Australian Research Council Linkage Project Grant LP110100569, which supported me during this research.

I would like to thank my parents for introducing me to the world of marine turtles and for their support through this project.

I would like to thank my children (Kiernan, Lachlan and Georgia) for providing the inspiration that anything is possible.

Lastly, I would like to thank my husband, Dr. Mark Flint, without whom this project would not have been completed. His invaluable support and faith in me allowed me to fulfil this project to its full extent.

Keywords

Marine turtle, Queensland, stranding, extreme weather event, rehabilitation, modelling, sea turtle

Australian and New Zealand Standard Research Classifications (ANZSRC)

ANZSRC code: 050211 Wildlife and Habitat Management, 65%

ANZSRC code: 060205 Marine and Estuarine Ecology (incl. Marine Ichthyology), 25%

ANZSRC code: 060809 Vertebrate Biology, 10%

Fields of Research (FoR) Classification

FoR code: 0602 Ecology, 40%

FoR code: 0501 Ecological Application, 35%

FoR code: 0707 Veterinary Sciences, 25%

Table of Contents

List of Figures

List of Tables

List of Abbreviations used in the thesis

AIC	Akaike's Information Criterion
CCL	curved carapace measurement
CI	Confidence interval
CITES	Convention on the International Trade in Endangered Species of Wild Fauna and Flora
cm	Centimetre
CMS	Convention for the Conservation of Migratory Species of Wild Animals
DAFF	Queensland Department of Agriculture, Fisheries and Forestry
EHP	Queensland Department of Environment and Heritage Protection
EPBC	*Environmental Protection and Biodiversity Conservation Act 1999*
GBRMPA	Great Barrier Reef Marine Park Authority
IUCN	International Union for Conservation of Nature and Natural Resources
MASH	Mobile Army Surgical Hospital
NCA	*Queensland's Nature Conservation Act 1992*.
p	Probability, statistical significance
qAIC	Quasi Akaike's Information Criterion
QLD	Queensland
QPWS	Queensland Parks and Wildlife
RWR	turtles which were released without rehabilitation and were then recaptured at a later date.
SCP	Shark Control Program
SOI	Southern Oscillation Index
TCL	tail to carapace length
TEDs	Turtle exclusion device
UQ	The University of Queensland
Vet-MARTI	Veterinary-Marine Animal Research, Teaching and Investigation

Chapter 1. Review of Literature

1.1. Rationale for marine turtle and extreme weather investigation

In recent years, subtropical regions such as Queensland have experienced many extreme weather events (Easterling et al., 2000a; Meehl et al., 2000b; Seneviratne et al., 2012), including snap freezes, droughts, cyclones and protracted rain depressions. Marine turtles have been proposed as sentinels of environmental health (Aguirre and Lutz, 2004; Hamann et al., 2010) and, as such, an increase in the numbers of animals which strand can indicate that the environments in which they live have changed.

In Queensland during the summer of 2010/2011, cyclones and protracted rain depressions caused wide-spread flooding which in turn led to increased periods of turbid water and increased nutrient and sediment loads from freshwater run-off being dumped into all four major coastal waterways (Brisbane, Fitzroy, Burnett and Burdekin Rivers) (Devlin et al., 2012a). The cyclones and floods stressed seagrass beds causing large scale die-off of ecologically important seagrass species and decreased water quality, intermittently along the entire length of the Queensland coastline south from Cairns (Coles et al., 2012; Great Barrier Reef Marine Park Authority, 2011a; McKenzie et al., 2014).

On average, around 500-800 marine turtles strand annually along the Queensland coastline (Biddle and Limpus, 2011). In 2011 there were over 1793 marine turtles reported stranded in Queensland in the Queensland Environment and Heritage Protection StrandNet database, of these 1408 were reports of dead or moribund turtles, and 385 were for reports of stranded animals which escaped unaided, were released after rehabilitation or were released *in-situ* (Meager and Limpus, 2012a). This was the largest annual number of turtles reported stranded in the 16 years for which comprehensive data has been collected for this region (Meager and Limpus, 2012a).

The flood events of 2010 and 2011 resulted in mass strandings. This raised a lot of public interest and action over rehabilitating turtles challenged by adverse weather events in an attempt to minimise the negative effect of the natural disaster and maximise the number of turtles that survived this catastrophic period.

It is known that within a year (short-term) of these types of catastrophes, marine megafauna show an increase in the number of stranding, mortalities and exacerbated by poor-health conditions (Meager and Limpus, 2014). However, the long-term (one or more years) and cumulative effect of all of these events on marine turtles is unknown. It is speculated to be detrimental to the survivorship of the local population. Rehabilitation of turtles found stranded along the coastline due to these conditions offers the potential to mitigate these negative effects on survivorship. For this project, survivorship is defined as being assessed at least once in the time after release from rehabilitation (usually by being observed during rodeo surveillance or during nesting beach surveys) and being found to be in good condition at each capture.

Marine turtles are seen as flagship species for conservation (Tisdell and Wilson, 2003) and thus their conservation is important for a variety of reasons particularly to protect ecological, aesthetic, economic, existence and bequest values (Aguirre and Lutz, 2004; Chaloupka et al., 2008b; Feck and Hamann, 2013; Jackson et al., 2001; Tisdell and Wilson, 2001). As such, community-based conservation efforts are increasing, as seen in the continual establishment of new marine turtle rehabilitation centres (Feck and Hamann, 2013).

Although efforts to conserve marine species in general, and improved knowledge of these cryptic marine reptiles, is required to ensure the long term viability of these populations (Hamann et al., 2010; Limpus and Chaloupka, 1997), the value and viability of current recovery strategies has been largely untested. This study adds to this knowledge base for marine turtles and provides information to enable management agencies and rehabilitation facilities to better their available strategies and resources.

1.2. Introduction

There are seven species of marine turtles found worldwide with the exception of the Polar Regions: green (*Chelonia mydas*), loggerhead (*Caretta caretta*), leatherback (*Dermochelys coriacea*), hawksbill (*Eretmochelys imbricata*), flatback (*Natator depressus*), olive ridley (*Lepidochelys olivacea*) and Kemp’s ridley (*Lepidochelys kempii*). With the exception of the Kemp’s ridley, all the other species occur as resident populations within Australian waters.

Direct human factors, such as marine debris, recreational and commercial fisheries by-catch, predation by introduced species, legal and illegal hunting of turtles and their eggs, coastal development (including beach armouring, beach nourishment, artificial lighting), recreational beach equipment, beach cleaning, beach erosion and boat strike, have been well documented to affect marine turtle morbidity and mortality (Alfaro-Shigueto et al., 2011; Bell et al., 2012; Lewison et al., 2014, 2004b; Limpus and Chaloupka, 1997; National Research Council Committee on Sea Turtle Conservation, 1990; Peckham et al., 2007; Wallace et al., 2010, 2008). What has not been comprehensively studied are the indirect human impacts, such as climate change and environmental stressors, which threaten turtles. This review assesses the relevant literature regarding the biology, ecology, historical research, veterinary treatment of marine turtles, rehabilitation of marine turtles, extreme weather events, climate change and critical habitat. Green, loggerhead and hawksbill turtles have been selected for this study due to their conservation status, stranding prevalence, historical and current research priorities and prevalence along the Queensland coast. The leatherback, flatback, olive ridley and Kemp's ridley were not considered for this study due to their low stranding and population numbers along the Queensland coastline, preventing a rigid, scientifically valid investigation of these species.

The information gathered during this review formed the basis to construct hypothesis, aims and objectives towards the relationship between strandings, extreme weather events and rehabilitation with respect to how they contribute to the management of marine turtle populations.

1.3. Turtle biology and ecology

1.3.1. Turtle evolution and history

Marine turtles belong to the Class Reptilia, Order Testudines and in terms of evolution appear to have not changed significantly in the last 110 M years (Environment Australia, 2003; Flint, 2010; Hirayama, 1998). Within Australian waters there are two surviving families of marine turtles, Cheloniidae (loggerheads, greens, olive ridleys, Kemp's ridleys, hawksbills and flatbacks) and Dermochelyidae (leatherbacks). There are common morphological features and life history traits between the two families including (Environment Australia, 2003):

- Most of their lives are spent in the marine environment with them needing to surface to breathe, and females must come ashore to lay white spherical eggs
- Long lived and generally slow to sexually mature and don't breed annually (Hamann et al., 2003)
- No teeth, but have keratinized sheathes covering beaks (Wyneken, 2003)
- No sense of taste but acute sense of smell (Moein Bartol and Musick, 2003)
- Colour vision with well-developed eyes (Moein Bartol and Musick, 2003)
- Low frequency hearing (Moein et al., 1993; Moein Bartol et al., 1999; Moein Bartol and Musick, 2003) .

1.3.2. Influence of Weather and Environmental factors on Turtles

It is known that weather and environmental factors influence the breeding status, timing of migration, incubation duration, embryonic sex determination, length of breeding season, breeding and interesting behaviour of many species of animal including marine turtles (Bowen et al., 2005; Cheng et al., 2009; Daunt et al., 2006; Doody et al., 2003; Hamann et al., 2003; Lamont and Fujisaki, 2014; Limpus et al., 1985; Sato et al., 1998; Seedang et al., 2008; Sexton et al., 1990; Standora and Spotila, 1985; Weber et al., 2012). Some of these are discussed below.

Numerous studies (Limpus, 2008a, 1989, Limpus and Nicholls, 2000, 1988) have reported that the proportion of adult female green turtles within a given feeding area preparing to breed is variable and is a function of the Southern Oscillation Index (SOI) two years prior to nesting. There have also been studies and reports which pointed out that males appeared to be regulated in a parallel manner (Limpus, 2008a, 1993; Limpus and Nicholls, 2000). It is thought that this regulation at a regional scale is impacted by the quantity and quality of forage (Broderick et al., 2001; Limpus, 2008a, 1993; Limpus and Nicholls, 2000).

Similarly, Limpus and Limpus (2003) showed that during any single year there is only a portion of adult female loggerhead turtles in a foraging area that are preparing to breed. However, they found that this is not mirrored during the same year across foraging areas such as Moreton Bay and Heron and Wistari Reef. This differs to the synchronicity seen with green turtles. No other environmental alternative factors have been able to provide a unifying explanation for the annual fluctuations in the loggerhead turtle breeding rates.

Marine turtles are ectothermic meaning that they are reliant on the surrounding environmental temperature, which can determine numerous biological and physical aspects of their life (Spotila et al., 1997). As such temperature is another factor that may affect our understanding of marine turtle ecology. Within Moreton Bay, loggerhead and green turtles are captured during winter on the intertidal banks where the water temperature has been recorded as low as 15 °C (Limpus and Limpus, 2003; Read et al., 1996). Through the ongoing monitoring studies of recaptured animals and satellite telemetry there is no evidence that east Australian loggerhead populations undertake north-south, summer-winter nonbreeding migrations leaving areas including Moreton Bay (Limpus and Limpus, 2003). This is in comparison to other populations of marine turtles which exhibit migration during cooler months (Carr and Caldwell, 1956; Musick et al., 1997; Witherington et al., 2006).

Temperature has also been shown to effect nest size, nesting frequency, nest (incubation success) and nesting success (Lamont and Fujisaki, 2014).

Temperature may affect the turtles' abilities to tolerate other potential co-occurring stressors, such as decreased food availability. The fact that numerous different age classes of turtles inhabit the same feeding area at the same time is an important fact in determining the susceptibility of different age classes to different environmental pressures and disease processes as there may be time lags evident in stranding rates.

1.3.3. Status of marine turtles in Australia.

Although climate has an impact on the breeding status of turtles on an annual basis, the threats faced by turtles vary within and between both species and populations as well as across temporal and spatial scales. Traditionally, these threats have mainly been attributed to anthropogenic sources (Feck and Hamann, 2013; Limpus, 2008a, 2008b).

Over the last several decades declining turtle populations have become cause for concern, following four centuries of harvesting, exploitation for eggs, meat, oils, leather, jewellery and ornaments (Campbell, 2003; Limpus, 2008a; National Research Council Committee on Sea Turtle Conservation, 1990). The majority of non-indigenous harvesting and trade of marine turtles world-wide has stopped due to the listing of turtles under the Convention for the Conservation of Migratory Species of Wild Animals (CMS) and the Convention on the

International Trade in Endangered Species of Wild Fauna and Flora (CITES) (Environment Australia, 2003). There still may be some illegal black trade of turtle products, all six species found within Australian waters are listed as threatened under the *Environmental Protection and Biodiversity Conservation Act 1999* and in Queensland waters under the *Queensland's Nature Conservation Act 1992*.

The status of different species and populations of marine turtles varies globally, but the World Conservation Union (IUCN) has recognised the overall decline of marine turtle populations and have listed all species on The Red List the except for the flatback which is classed as data deficient.

Species	**Queensland - NCA**	**Australia - EPBC**	**IUCN**	**CMS**	**CITES**
Loggerhead	Endangered	Endangered	Vulnerable	I and II	Appendix I
Green	Vulnerable	Vulnerable	Endangered	I and II	Appendix I
Leatherback	Endangered	Vulnerable	Vulnerable	I and II	Appendix I
Flatback	Vulnerable	Vulnerable	Data deficient	II only	Appendix I
Hawksbill	Vulnerable	Vulnerable	Critically Endangered	I and II	Appendix I
Olive ridley	Endangered	Endangered	Vulnerable	I and II	Appendix I
Kemps ridley	NA	NA	Critically Endangered	I and II	Appendix I

Worldwide, conservation of turtles seems to be having a varied impact. It has been suggested that the southern Great Barrier Reef green turtle population increased significantly between 1985 and 1992. This increase occurred at a rate of approximately 10.6% per annum (Chaloupka and Limpus, 2001). As a contrast the southern Great Barrier Reef foraging loggerhead turtle populations has declined over this same period at an approximate rate of 3% pa (Chaloupka and Limpus, 2001). Green turtle conservation in the Hawaiian Islands has resulted in the sub-population being delisted in recent years.

1.3.4. Ecology of turtles

In Australia, young green and loggerhead turtles recruit to their benthic feeding ground when they reach approximately 40-50 cm (curved carapace measurement (CCL)) (Limpus, 2008a; Limpus and Chaloupka, 1997; Limpus and Limpus, 2003) and remain in that area for extended periods of time (years to decades). After this, immature and adult green and loggerhead turtles feed in tidal and sub-tidal habitats including coral and rocky reefs, seagrass meadows and algal turfs on sand and mud flats within the continental shelf bounded by the East Arafura Sea, Gulf of Carpentaria, Torres Strait, Gulf of Papua, Coral Sea, Great Barrier Reef, Hervey Bay, Moreton Bay and New South Wales coastal waters (Limpus, 2008a; Limpus and Reed, 1985a; Read and Limpus, 2002; Speirs, 2002). Based on tag recoveries of adults the majority of the southern Great Barrier Reef green turtle stock occupies feeding areas to the south of Princess Charlotte Bay to northern New South Wales and New Caledonia (Limpus, 2008a; Limpus et al., 2003).

Green turtles represent the largest proportion of the Queensland marine turtle populations and small immature animals are the largest cohort of this population (Chaloupka, 2002a; Chaloupka and Limpus, 2001).Within coastal waters green turtles are almost exclusively herbivorous, feeding principally on seagrass, a wide range of algae and mangrove fruits (Brand-Gardner et al., 1999; Limpus, 2008a; Read and Limpus, 2002). Occasionally, green turtles feed on macroplankton, including jellyfish, bluebottles, small crustaceans and dead fish (Limpus, 2008a; Read and Limpus, 2002). Brand-Gardner et al.(1999) found that within Moreton Bay small immature green turtles forage selectively on plants with higher nitrogen levels and lower levels of fibre (such as *Gracilaria* sp.). This makes them susceptible to starvation when there are decreases in seagrass coverage. The foraging grounds and coastal habitats used by greens globally are at risk from human settlement and coastal land development (McKenzie et al., 2010; National Research Council Committee on Sea Turtle Conservation, 1990; Waycott et al., 2009). Further, small immature turtles are likely to be the most susceptible cohort to these and other threats, due to having a naïve immune system to numerous potential stressors and being obligate residents of nearshore habitats that may be subject to a range of environmental stressors (Flint et al., 2010c, 2010d, Limpus et al., 2007, 1994a).

Numerous studies have found that when adults are in breeding condition they make migrations to traditional breeding sites, and at the completion of the breeding season they return to the same feeding area (Avens et al., 2003; Broderick et al., 2007; Hawkes et al., 2012; Limpus et al., 1992; Marcovaldi et al., 2010; Musick et al., 1997; Shimada et al., 2016b; Vélez-Rubio et al., 2013; Watanabe et al., 2011). Some of these animals have been recorded migrating over 2600km between feeding areas and breeding sites (Limpus, 2008a; Limpus et al., 1992). This displays the wide range of habitats that turtle’s use and the fidelity that they exhibit to these sites (Limpus, 2008a; Limpus et al., 1992). When the breeding animals make these physiologically-demanding migrations, studies have suggested that feed intake is greatly reduced or totally absent, particularly for females during egg production (Hamann et al., 2003, 2002; Jessop et al., 2004; Kwan, 1994; Limpus, 2008a, 1993; Limpus et al., 2001; Tucker and Read, 2001). This reduction in feeding in addition to the physiological challenges of migration has the potential to increase the susceptibility to disease, which is exacerbated if forage has been decimated upon their return to the feeding area.

When looking at the timing of these breeding migrations, Limpus & Limpus (2001, 2003) found that adult female loggerhead from the southern Great Barrier Reef foraging areas (23°S) commence their breeding migrations in late October to early November, whereas females from Moreton Bay (27°S) depart mid-November. The different timing of these may affect when adults are exposed to different/increased threats.

In comparison, within the coastal waters of eastern Australian, loggerheads are carnivorous, feeding on hard-bodied slow moving invertebrate pray including gastropod, bivalve molluscs, portunid crabs and hermit crabs. They feed less frequently on softer bodied invertebrates including jellyfish, anemones, holothurians, sea urchins and fish (Limpus, 2008b). This difference in food preference may delay the impact weather-related food availability compared to herbivorous turtle species such as greens.

1.3.5. Current Research programs (or activities)

The life history stages of marine turtles (eg. long distance migrations and use of various habitats) make it difficult to assess biological and population parameters (Komoroske et al., 2017). As such researchers need to use a wide range techniques (Wyneken et al., 2013). Some of these techniques related to this thesis are discussed below.

The Queensland Department of Environment and Heritage Protection has a current research program which has been ongoing since 1968 (Limpus and Limpus, 2003). This program has four major elements:

- Monitoring (tagging census and stranding database);
- Research (demographic studies at nesting beaches and feeding areas, population genetics studies, migration studies, incubation/embryological research, ENSO

regulation of green turtle breeding rates, nutritional studies, health studies and population modelling); and

- Management (fox baiting to improve loggerhead breeding success and environmental education programs)(Environment Australia, 2003).
- Habitat protection (nesting beaches in National Parks; foraging, inter-nesting habitats in Marine Protected Areas)

The continuation of a monitoring program similar to the one in Queensland is the key to managing marine turtle populations (Environment Australia, 2003). It is necessary to determine the status of marine turtles and to detect changes in populations and also measure the effectiveness of management actions (Environment Australia, 2003). The tagging program that is undertaken in Queensland has provided much of the current knowledge about marine turtle behaviour and ecology but is not without its recognised limitations (Environment Australia, 2003). The major downsides of this type of program on a large scale (Queensland Coastline) is that it can take many years before a turtle is recaptured and decades to build a database about migration destinations; hundreds or thousands of animals are tagged but this only yields few returns if the target site is non-selected, the success of which can rely on the initiative, interest and understanding of the person capturing the turtle (Environment Australia, 2003).

1.3.5.1. *Satellite Tracking*

Satellite tracking can overcome some of the short-falls of the Queensland monitoring program and provide data on the movement behaviour, migration routes and locations of potential habitats. Overall satellite tracking has much to offer but the use of this method as anything other than studying individual movement behaviour is dependent on achieving sufficiently large sample size to make robust hypotheses (Cardona et al., 2012; Environment Australia, 2003; Godley et al., 2008; Hays, 2014).

Shimada et al. (2016) analysed satellite tracking data of animals which had been displaced from their original capture site (inferred home area). Of the 59 displaced turtles, 52 returned to their home areas. All 52 non-displaced turtles remained within their home areas. This indicates that animals which are removed from their home area are likely to return to that location and be exposed to the same threat/conditions as they were previously.

Mestre et al. (2014) conducted satellite tracking of turtles that were released from rehabilitation after prolonged amounts of time in captivity. The animals used during this study exhibited a targeted directed movement towards recognised feeding grounds of each species, which could be indicative that they were returning ‘home’, further supporting arguments by (Shimada et al., 2016b). This study only tracked animals for an average of 688 days, although 2 years of data is informative, considering that marine turtles are long lived and there are long time frames where turtles are unobserved, this is a relatively short period.

Schofield et al. (2008) satellite tracked a Harbor Porpoise that had been undergoing rehabilitation for approximately 10 months. The animal was released 2880 km from its original stranding location in a site that was considered suitable habitat. After 63 days the animal returned to a location within several kilometres of its original stranding location.

Bellido et al. (2010) discussed an animal that displayed abnornal behaviour after being released from rehabilitation 14 months after being admitted. The behaviour noted during this study was an example of habituation, where the animal was reported approaching people.

1.3.5.2. *Capture-Mark-Recapture Program*

Numerous studies have discussed the variation among feeding grounds of marine turtle recapture rates, some of these are discussed below. Bell et al.(2012) reported on an 11-year capture-mark-recapture program of Hawksbills, in the far northern section of the Great Barrier Reef Marine Park. This study had varying percentages of recaptures with the lowest occurring the year after the project commenced (1.4%) and the highest occurring 8 years (32.4%). After this year, the percentages varied between 19.2 and 30.5%.

Chaloupka and Limpus (2002) reported on a capture-mark-recapture program over a 9-year program, with 36% of all animals caught only once, 14% caught twice, 14% caught three times and 36% captured at least four times. When examining the mean annual survivorship of adults and immature turtles they found no significant differences. However, they also found large numbers of animals that were only captured originally and then not seen again.

Chaloupka and Limpus (2002) analysed mark – recapture studies conducted in the southern Great Barrier Reef coral reefs and found that there was no sex-specific difference in the annual survival probability for immature or adult loggerhead turtles. This means that there should be no differences in the stranding rate of immature or adult loggerhead turtles. If there is a significant difference this could indicate that there is an increased impact occurring on that sex or age class.

Based on capture-mark-recapture programs, Chaloupka (2002b) estimated the total southern Great Barrier Reef benthic green turtle population to be 641 262. Chaloupka and Limpus (2001), also noted that during the period between 1985 and 1992 that the southern Great Barrier Reef resident green turtle population has increased at approximately 11% per year. Chaloupka et al. (2008a) and Chaloupka and Limpus (2001) also estimate that the nesting female population of green turtles in the southern Great Barrier Reef is increasing at approximately 3% per annum.

The recapture of juveniles and adults at feeding grounds provides valuable data on growth, population size and structure (Environment Australia, 2003). The recapture of tagged turtles at places other than site of original capture provides information on the distance travelled and potential locations of nesting and foraging habitats (Environment Australia, 2003).

1.3.5.3. *Genetic Analysis*

Genetic analysis has been used to investigate natal homing, establish support for connectivity between foraging and nesting areas as well as revealing population structure (Jensen et al., 2013; Komoroske et al., 2017). Genetic analysis has helped management agencies to define management units (eg Evolutionary Significant Units (ESUs), Distinct Population Segments (DPSs) and Regional Management Units (RMUs) (Jensen et al., 2013; Komoroske et al., 2017). Genetic analysis has also highlighted the fact that different populations consist of animals from different distances away, for example in Australia animals come from the nearest genetic stock, whereas places such as New Caledonia, Colombia and Japan has animals from distance stocks, as far as 2000 km away (references within (Komoroske et al., 2017).

While numerous threats to marine turtles are obvious, one such threat which is not evident is the loss of genetic diversity (Komoroske et al., 2017). The use of genetic analysis enables researchers and managers to assess this threat and implement measures to mitigate this threat (Komoroske et al., 2017)

1.3.6. Diseases

Until recently marine turtle disease have remained largely unstudied. The majority of disease investigations to date in marine turtles has focused on either generalised disease syndromes effecting animals at the population level or specific disease that have been studied very closely (eg spirorchiidiasis and fibropapillomatosis). This has provided insight to the range of conditions which are impacting functional populations but has left gaps in the knowledge base on the true effects of disease on turtles.

However it has been shown that the analysis of stranding data over wide spatio-temporal ranges can provide insight into disease, geographic ranges, seasonal distribution and life history of both the local and larger population (Balazs, 1991; Herbst, 1994; Scherer et al., 2014).

Some animals may become stranded due to visual anthropogenic impacts but there may be underlying disease processes occurring which is making turtles more susceptible to things such as boat strikes (Environment Australia, 2003).These underlying disease processes are only beginning to be understood and there is still more knowledge to be gained. Within StrandNet, the disease category of mortality was characterized by poor to very poor body condition during external examination (in the absence of a necropsy being performed) or by diagnosis subsequent to an internal examination (necropsy). The aetiology of the recorded natural causes of death is predominantly unknown. It is suspected that predisposing health factors or subclinical diseases may have been exacerbated due to an underlying environmental problem (Meager and Limpus, 2012a).

One of the most recent disease outbreaks which has being studied is spirorchiidiasis (Aguirre et al., 1998; Chapman et al., 2015; Flint et al., 2010a; Glazebrook et al., 1989; Glazebrook and Campbell, 1981; Gordon et al., 1998). The prevalence of this disease has had varying influence, but it is thought to infect between 75% and 98% of turtles based on turtles presenting for necropsy having spirorchiid or lesions caused by them. In terms of it causing death, in 41% of cases examined by Flint et al. (2010c), spirorchiids were a contributor to the cause of death. Although there have been great advances made in the last 5 years towards developing a “pool-side” test for animals admitted to rehabilitation, this diagnostic tool has not been finalised (Chapman, 2017)

Another disease process which has been identified to cause death in marine turtles is coccidiosis (Flint, 2010; Gordon, 2005; Gordon et al., 1993). There has been on-going records of this disease epidemic occurring throughout Queensland between 1990 and the present day (Flint, 2010; Gordon, 2005; Gordon et al., 1993). There have been recent advances towards developing an ante-mortem test for this disease syndrome to better respond to outbreaks (Chapman et al., 2016).

An additional disease which has been occurring for a long time in marine turtles is fibropapillomatosis. First described in marine turtles in 1938 (Smith and Coates, 1938), marine turtle fibropapillomatosis is characterised by cutaneous masses, which are apparently infectious between animals (Aguirre and Lutz, 2004; Chaloupka et al., 2009; Flint, 2010; Flint et al., 2010b; Herbst, 1994; Herbst et al., 1999). The masses associated with fibropapillomatosis are benign and are generally not directly associated with cause of death (Landsberg et al., 1999). However, if they get large enough they can cause problems to mechanical process associated with swimming, diving, location and capture of prey (Aguirre and Lutz, 2004; Flint, 2010; Flint et al., 2010b; Herbst, 1994; Landsberg et al., 1999). It has also been noted that turtles with multiple fibropapillomas can become visually debilitated with blood chemistry and cell counts supporting this observation (Herbst, 1994; Norton et al., 1990)

Chaloupka et al., (2008b) reported that fibropapillomatosis was the most common known cause of stranding in Hawaiian waters. In Hawaii, the rate at which green turtles stranded due to fibropapillomatosis increased from 1982 but levelled off during the mid-1990s (Chaloupka et al., 2008b). The increase in fibropapillomatosis occurring worldwide and the spread of it to areas where it has not previously been recorded makes it one of the most significant diseases of reptiles (Herbst, 1994).

1.4. Aquatic Vegetation and Coral

Seagrass beds, coral reefs, mangroves and other inshore ecosystems provide essential habitat for many species such as fish, turtles and dugongs and facilitate critical environmental and biological processes (Bell and Lovelock, 2013; Costanza et al., 1997; Great Barrier Reef Marine Park Authority, 2011a; Jackson et al., 2001; Ogden et al., 1983; Olds et al., 2014; Polidoro et al., 2010; Schaffelke et al., 2005; Short and Wyllie-Echeverria, 1996; Thayer et al., 1982, 1984; Waycott et al., 2005; York et al., 2016).

However, despite their importance, mangroves are being removed from populated estuaries within the Great Barrier Reef region to make way for coastal development (Polidoro et al., 2010; Schaffelke et al., 2005); coral reefs are receding under climate change (Brodie et al., 2012; Brodie and Waterhouse, 2012; Great Barrier Reef Marine Park Authority, 2015; Haward et al., 2013; Steffen et al., 2014); and pollution is impairing seagrass beds (Brodie et al., 2012; Great Barrier Reef Marine Park Authority, 2015; Joo et al., 2012; Waycott et al., 2009).

Changes in water quality affect all the marine plant communities which occur in the inshore environments (Schaffelke et al., 2005). Due to their proximity to land based point sources, mangroves are the most exposed community types (Schaffelke et al., 2005). Mangroves also assist other communities to deal with increased sediment, nutrient and pollutant loads by acting as filters and traps (Schaffelke et al., 2005).

Seagrass, mangrove, coral reef and coastal wetland habitat losses have been reported worldwide and as such cumulatively these losses are signalling a concerning deterioration in all nearshore environments (Waycott et al., 2009).

The Great Barrier Reef Marine Park, covers over 247800km^2 , of which only 6% of the region is covered by coral reefs while the shallow inter-reef and lagoon areas cover 58% (Coles et al., 2015; Wachenfeld et al., 1998). As such they form the basis of the ecosystem analysis which follows.

1.4.1. Seagrass

In terms of seagrass biodiversity, tropical and sub-tropical Australia has one of the richest areas in the world (Environment Australia, 2003; Waycott et al., 2005; York et al., 2016).

Seagrass communities provide essential habitats for several different species of animals (Coles et al., 2007; Jackson et al., 2001; Ogden et al., 1983; Short et al., 2014; Short and Wyllie-Echeverria, 1996; Thayer et al., 1984, 1982; Waycott et al., 2005; York et al., 2016). Within the Great Barrier Reef healthy seagrass meadows provide habitat for numerous invertebrates, fish and algal species as well as being an important food resources for dugongs, green turtles and numerous commercially important fish species (McKenzie et al., 2012; Short et al., 2014; Waycott et al., 2005).

1.4.1.1. *Seagrass decline*

There are several known causes of seagrass loss including sewage outfalls, dredging, dugong over-grazing, boat traffic and flooding (Referecnes in Preen et al., 1995; Short and Wyllie-Echeverria, 1996). These also cause overall environmental health degradation. Although seagrass loss can be directly linked to green turtle mortality, the causes of seagrass loss can also be linked to decreases in food availability to loggerheads and other species (Heck et al., 2008).

Australia, as has occurred in many regions throughout the world, has experienced large scale seagrass meadow disturbances and losses over the last several decades (Waycott et al., 2009). This has occurred on both large and small scales with the extent of loss and timing of loss being determined by the duration, frequency and type of disturbance (Campbell and McKenzie, 2004). The losses can occur at spatial and temporal scales that can be due to man-made or natural causes and in often cases interactions between the two (Coles et al., 2015; Environment Australia, 2003; Preen, 1995). It has been suggested that seagrass is being lost at a rate of 7% per year or approximately $100km^2$ yr^{-1} (Waycott et al., 2009).

Seagrass decline can be impacted by seasonality, with loss of meadows during summer having a greater annual impact than losses suffered during winter. The seasonality of seagrass die off has been studied by Kerr and Strother (1990); Lanyon and Marsh (1995) and Mellors et al. (1993). Kerr and Strother (1990) found above ground biomass of *Zotera muelleri* to be at a minimum in the winter months (particularly June to August), while the maximum biomass occurred during the summer months (October to February). Lanyon and Marsh (1995) found similar results with total seagrass abundance and individual species fluctuated seasonally with die-offs occurring during August to September, and recovery occurring between November and March. Mellors et al. (1993) found slightly different results, with die-offs occurring between May and August with recovery occurring September to December. Mellors et al. (1993) also noted some inter annual variability with when die-offs occurred.

There have been ongoing declines of seagrass communities reported in Moreton Bay. These have been attributed to the deterioration of water quality linked to urbanisation, industrialisation and increased land use which have all resulted in an increase in nutrient loading, sedimentation and influx of contaminants and toxins as well as other detrimental effects on seagrass communities (Abal and Dennison, 1996; Environment Australia, 2003; Hyland et al., 1989; Kirkman, 1978; Short and Wyllie-Echeverria, 1996).

The identification that land use practices impact turtles in eastern Australia is a cause for concern. These practices have been identified as land clearing, urban and industrial development (Brodie et al., 2011; Environment Australia, 2003; Short and Wyllie-Echeverria, 1996). Better management of catchments, urban runoff, effluent and discharges can improve the water quality thereby improving the quality of seagrass meadows and the reduction of algal growth (Abal and Dennison, 1996; Environment Australia, 2003). The land clearing of coastal areas for residential or industrial development has the potential to affect turtle populations in multiple ways during various times of their life cycle (Environment Australia, 2003). Coastal development brings additional impacts including increased run-off from paved areas, increased turbidity in water and increased levels of chemicals (Environment Australia, 2003). Increases in sewage discharge may increase nutrient loads, particularly levels of phosphates and also encourage algal growth (Environment Australia, 2003). All of these land use practices can be amplified by the increase in run off produced by flooding events.

1.4.1.2. *Seagrass recovery*

Recovery rates of seagrass meadows depend on how much damage has occurred, meadows with intact seed banks or remnant plants displayed strong recovery 12 month after the disturbance where as other slower-growing species and areas with diminished seed banks may not recover for decades (Great Barrier Reef Marine Park Authority, 2011a).

Restoring the natural resilience of important habitats is more important now than ever before as increased flooding and more severe storms occur (Great Barrier Reef Marine Park Authority, 2011a). Different tropical seagrass species have shown different recovery rates and methods post large-scale climatic disturbances. This was shown during Rasheed et al. (2014) study in which reproductive strategy and the presence of viable seed banks influenced whether recovery occurred or not. The natural recovery of seagrass meadows depends on interactions between light availability, nutrient loads and the availability of recruits, including seeds and any remaining propagules (McKenzie et al., 2014, 2012).

1.4.1.3. *Seagrass and water quality*

There are numerous factors which affect the water quality of water discharged on the Great Barrier Reef, these include land-based runoff and river flow, point source pollution and extreme weather conditions (Brodie et al., 2011; Waterhouse et al., 2012). Decreased water quality parameter have been identified as one of the most significant causes of seagrass decline (Brodie et al., 2012; Coles et al., 2015; Day and Dobbs, 2013; Grech et al., 2012, 2011, McKenzie et al., 2014, 2012).

The large flood events associated with tropical cyclones and monsoonal rainfall are the dominate mechanisms associated with wet and dry tropic river system discharge (Devlin and Schaffelke, 2009). The Wet Tropics regions spans from Cooktown in the north to Townsville in the south (Turton, 2005). The Dry Tropics region spans from Townsville south to Mackay (Herr et al., 2004). Within the Wet Tropics most of the rivers flow into small catchments that are characterised by low inter-annual variability of rainfall with multiple short-duration flow events each year. This contrasts to the Dry Tropics where discharge occurs as one or two small annual flows, or occasionally as very large flood event greatly exceeding other regional rivers and lasting for several weeks (Devlin et al., 2012a).

The Burdekin, Mackay Whitsunday and Burnett Mary regions are of the greatest concern in regards to seagrass loss, both in regards to abundance but also recovery. Previous studies have shown that there are very poor seed banks and reproductive effort occurring (McKenzie et al., 2012).

1.5. Extreme Weather Events

During summer in Australia there is a heightened risk to extreme weather and warmer temperatures. Summer brings warmer conditions which increases the threat from higher sea temperatures, the wet season also brings strong monsoon conditions which can result in large flooding plumes and damaging cyclones (Great Barrier Reef Marine Park Authority, 2011a).

Regional seasonal weather conditions such as the El Niño Southern Oscillation cause changes in the underlying weather conditions, either being clear and dry (El Niño) or wet and cloudy (La Niña). During La Niña events more damage is caused by flood and cyclones (Great Barrier Reef Marine Park Authority, 2011a).

Flooding is a periodical natural event with impacts occurring at numerous scales (Howes et al., 2013). Over recent years the frequency, duration and intensity of extreme rainfall events has changed. The weather conditions currently being experienced have not been recorded in the history of the Great Barrier Reef Marine Park (Great Barrier Reef Marine Park Authority, 2011a). This includes increase in the frequency of cyclones, increases and decreases of rainfall (different years and regions) and more extreme temperatures.

1.5.1. Extreme Weather and Marine Turtles

Green turtles are almost entirely reliant on seagrass, algae and a wide range of mangrove fruits for nutrition (Arthur et al., 2009; Bjorndal, 1997; Brand-Gardner et al., 1999; Limpus, 2008a; Limpus et al., 2005). Due to this strong dependency on aquatic vegetation, green turtles that live within inshore coast habitats where seagrasses are a large component of their diet have suffered during and were found in poor condition post the extreme weather of 2010-11. It is likely they are partially able to compensate for the decreased seagrass availability for several weeks to a few months by eating algae and mangrove leaves and having relatively low energy demands. Although these lower nutritional value foods render them more susceptible to ill health and death (Great Barrier Reef Marine Park Authority, 2011a).

Meager and Limpus (2012) stated that the most plausible explanation for the high rate of strandings and mortalities of near shore green turtles during 2011 was extreme weather events that occurred in late 2010 and early 2011, which impacted on seagrass foraging areas. They linked this because most of the examined mortalities were attributed to protracted ill health/poor body condition in green turtles and dugongs; which both primarily forage on seagrass (Marsh and Kwan, 2008). There was evidence that seagrass pastures in Queensland were impacted by elevated rainfall, flooding and a cyclone during the summer of 2010/2011 (Coles et al., 2015; McKenzie et al., 2014, 2000). They also stated that elevated rates of turtle mortalities have occurred following similar weather events in the past (Meager and Limpus, 2012a).

Meager and Limpus (2014), found links between periods of elevated freshwater discharge, low air temperatures and increased dugong mortality. They found that 9 months after elevated freshwater discharge there was an increase in dugong mortality.

1.5.2. Impact of extreme weather on aquatic vegetation and coral

Extreme weather events can impact seagrass beds, mangrove forests and coral reef and other inshore ecosystems (Bell and Lovelock, 2013; Crow, 2011; Great Barrier Reef Marine Park Authority, 2011a). The impacts experienced by these important ecosystems have flow effects through the entire systems to species that depend on them for food and shelter (Great Barrier Reef Marine Park Authority, 2011a).

The impacts that these ecosystems face is varied depending on the system and exposure. Coral reefs can be impacted by damaging turbulence, destructive currents and flood plumes (Great Barrier Reef Marine Park Authority, 2011a). Seagrasses can be impacted by flood plumes, damaging wave action and currents (Cardoso et al., 2008; Great Barrier Reef Marine Park Authority, 2011a; Preen and Marsh, 1995; Preen et al., 1995). While other inshore ecosystems can be affected through strong winds and unusual and/or prolonged inundation (Bell and Lovelock, 2013; Great Barrier Reef Marine Park Authority, 2011a).

Flood waters entering the marine environment carry with them increased sediment, chemical and pesticide loads which can all impact on the near shore environments (Devlin et al., 2012a, 2012b; Great Barrier Reef Marine Park Authority, 2011a).

Coral can be directly impacted upon by wave action breaking corals and also indirectly which may take several years to be fully understood (Devlin et al., 2012a; Great Barrier

Reef Marine Park Authority, 2011a). Studies have shown that between 1995 and 2009 approximately 34% of all coral morality on the Great Barrier Reef during the long term-monitoring programs can be attributed to storm damage (Great Barrier Reef Marine Park Authority, 2011a).

While extreme weather events have a negative effect on coral reefs and seagrass, they also can cause increases in macroalgae growth (Crow, 2011; Great Barrier Reef Marine Park Authority, 2011a; Schaffelke et al., 2005).

1.5.2.1. *Extreme Weather and Seagrass decline*

Historically, numerous large scale seagrass die-offs have occurred which most can be attributed to flooding events locally (Poiner et al., 1993a; Preen et al., 1995). In 1985 Tropical Cyclone Sandy caused over 183km^2 of seagrass loss in the Gulf of Carpentaria. This equated to about 20% of the seagrass at the time, but after 12 years much of the area had recovered; however there was still a large area about 20km^2 that was devoid of seagrass (Poiner et al., 1993a). In 1992-1993 there was an estimated 900km^2 of seagrass in Hervey Bay that disappeared, the cause of which is unknown, but it's thought to be linked to high turbidities resulting from flooding (Preen et al., 1995). There has also been 1199km^2 of seagrass loss in Torres Strait, it is suspected that this was also the result of high turbidities as a result of flooding of the Mai River (Long et al., 1997).

During their studies Campbell and McKenzie (2004) looked at the effect of flooding on the timing of seed germination during the initials stages of recovery and the influence that water quality plays on this processes. The key finding of this study was that within 2 years of a flood-related loss, sub-tropical seagrass populations returned to pre-flood abundances. Our work has shown that 8 months after a loss of seagrass like this, marine turtles begin to succumb to inanition and secondary conditions and strand. This implies the impact can carry on for long after the initial stranding response. This process involved 2 phases seeding growth (initials germination) occurring 18 months post flood and then full growth recover 6-9 months after that. The time interval that is required after severe seagrass loss and the ability to form meadows after disturbances is influenced mainly by the light quality, although nutrient availability and sediment characteristics are likely to promote seed germination if conditions are favourable (Campbell and McKenzie, 2004; Environment Australia, 2003). Campbell and McKenzie (2004) also found site variations in the onset of seedling growth, they suggested that seedling growth following large scale seagrass loss may also depend on physical and chemical characteristics of sediments and the water.

The most likely cause of seagrass loss post flood event was decreased light availability caused by high concentration of sediments and nutrients, Campbell and McKenzie (2004) as short-term increases in turbidity is known to inhibit seagrass photosynthesis, which in turn affects carbohydrate concentrations thus altering leaf and rhizome growth. Different species of seagrass survive for different periods of time below minimum light availability, smaller species that can only store small amounts carbohydrates survive for shorter periods of time. The difference in survival of seagrass species can also create time lags as animals with different diet preferences become susceptible to decreased or changed food availability.

The rate of seagrass decline depends on several factors as does their recovery rates. The rate of seagrass declines depends on the type of seagrass community, some species are able to tolerate longer periods of light limitations than other species (Great Barrier Reef Marine Park Authority, 2011a).

Preliminary survey results of seagrass meadows in southern Great Barrier Reef indicate that extensive and prolonged floods have caused significant damage (Rasheed et al., 2014). There are indicators that many shallow water or intertidal meadows have suffered severe scouring within the area affect by gale force winds. Deep water surveys also indicated that seagrass meadows down to at least 30m have been found almost completely barren following cyclone Yasi (Great Barrier Reef Marine Park Authority, 2011a). Subsequent studies have found that recovery is based on life history traits and species at the location (Rasheed et al., 2014)

Following the flooding of the Mary River in February 1999, approximately 90% of the intertidal seagrass in the northern Great Sandy Strait disappeared, by 2002 the seagrass cover returned to the pre-disturbance amount (Campbell and McKenzie, 2004).

The high turbidity observed for less than 30 days following the 1999 flood of the Mary River and resultant large scale loss suggested to Campbell and McKenzie (2004) that light reduction alone cannot fully explain seagrass die-off. Other factors which contribute to seagrass die-off include sediment deposition, sediment disturbance, and salinity reduction.

Coles et al. (2012) reported that up to 2009 seagrass meadows between Torres Strait and Hervey Bay were mostly stable until tropical cyclones Larry and Yasi and the severe floods of 2011 which resulted in the loss of coverage and abundance. They also found there was regional variation in seagrass abundance and the impacts faced with the leading threats to coastal seagrasses being terrestrial based.

There was evidence that seagrasses in Moreton Bay and Hervey Bay were impacted upon by flooding and/or high levels of river discharge in Brisbane, Burnett and Mary Rivers (Meager and Limpus, 2012a).

Prior to the events of 2010/2011, many of the intertidal seagrass meadows had shown a trend in declining abundance. This data indicates that seagrasses and the species which rely on them are especially vulnerable to the changing conditions and require increased management focus in coming years. It is likely that the impacts of the extreme weather events of 2010/2011 exacerbated the longer-term decline of seagrass abundance that had been observed (Great Barrier Reef Marine Park Authority, 2011a; McKenzie et al., 2012).

1.5.3. Occurrences/frequency

Between 1970 and 2006 the Great Barrier Reef has been exposed to 116 cyclones and the associated damaging winds, with the frequency of severe cyclones during the last three decades almost doubling (Great Barrier Reef Marine Park Authority, 2011b; Webster, 2005).

There was a strong La Niña event in mid-2010 which resulted in elevated rainfall across the Queensland coast with severe floods in southern and central Queensland from December 2010 through to January 2011 making it the second wettest summer on record (Devlin et al., 2012a; Meager and Limpus, 2012a). During the 2010-2011 summer 3 tropical cyclones crossed the Northern Queensland Coast. In December, tropical cyclone Tasha crossed the coast near Innisfail, and moved south causing severe flooding in the Brisbane, Burnett, Fitzroy and Burdekin Rivers (Devlin et al., 2012a). Following this tropical cyclone Anthony crossed the coast near the Whitsundays as a category 2, and moved inland causing flooding in southern Australia (Devlin et al., 2012a). During 2011 tropical cyclone Yasi crossed the northern Queensland Coast near Cardwell which resulted in extensive seagrass loss in the Missionary Bay/Hinchinbrook area and Cleveland Bay (Devlin et al., 2012a; Meager and Limpus, 2012a).

As a result of tropical cyclone Yasi extensive damage was recorded to coral and seagrass within a 300 km wide area across the continental shelf (Great Barrier Reef Marine Park Authority, 2011b). A consequence of tropical cyclone Yasi's destructive winds was estimated that approximately 98% of the intertidal seagrass within the marine park was lost (McKenzie et al., 2014).

The impacts to seagrass from tropical cyclone Yasi over the area from Hervey Bay to Cairns, compounded on the reports of declining seagrass health since 2009 Mckenzie et al. (2014). The impacts to seagrass communities prior to 2011 were shown by Mckenzie et al. (2014) to included increased mortality and decreased coverage, and were exacerbated by long periods of low salinity, smothering by sediment and reduced light availability which were associated with the extreme weather events.

The wet season of 2010/2011 started early, with the Wet Tropics reporting high flows during November and December 2010 with the season continuing into April (Devlin et al., 2012a). Various levels of flooding was observed in one or more part of the Great Barrier Reef for the 4 month period (Devlin et al., 2012a). During this summer there was a persistent flood plume observed out from the Burdekin, Fitzroy, Burnett and Mary Rivers. The Great Barrier Reef experienced one of the most powerful cyclones since records began, south east Queensland also recorded up to 400% higher rainfall than normal (Great Barrier Reef Marine Park Authority, 2011a).

1.5.4. Southern Oscillation Index

The Southern Oscillation Index (SOI) is used to illustrate the relationship between surface pressure, temperature and precipitation (Rasmusson and Carpenter, 1982). The sea level pressure difference between Tahiti and Darwin are used to calculate the SOI (Hoyos et al., 2013; McBride et al., 2003; Nicholls et al., 1997; Rasmusson and Carpenter, 1982). In turn the SOI is then used to indicate whether an El Niño or La Niña will develop in the Pacific Ocean and its intensity (McBride and Nicholls, 1983). As such it can be used as a predictor to determine the rainfall levels expected and extreme values of the oscillation can cause extreme weather events to occur (McBride et al., 2003; McBride and Nicholls, 1983).

1.5.5. Flood Plumes

Within the Great Barrier Reef, river-run off is the principle carrier of sediment, nutrients, pesticides and chemical pollutants (Devlin and Schaffelke, 2009; Fabricius et al., 2012; Katharina E Fabricius, 2005; Furnas, 2003). This run-off mainly occurs during the 5-month summer wet season (Devlin and Schaffelke, 2009; Furnas, 2003). These events have always occurred, but over the last 200-years, changes in land use have resulted in increased levels of nutrients, suspended sediments and pesticides which are now having increased impacts on coastal and inshore environments (Brodie and Mitchell, 2005; Devlin and Schaffelke, 2009; Katharina E. Fabricius, 2005; Furnas, 2003; Schaffelke et al., 2005; Waycott et al., 2005). It is known that increased turbidity, boating traffic and dredging activities, effluent discharge, eutrophication and increased herbicide concentrations have a negative effect on the growth and abundance of seagrasses in inshore and coastal ecosystems (Cuttriss et al., 2013; Devlin and Schaffelke, 2009; Schaffelke et al., 2005; Waycott et al., 2005).

1.6. Climate Change

In the last century there have been four category 5 cyclones which have affected the reef compared to the two last centuries, which both occurred in 1918. Climate scientists believe that the increase in frequency of Extreme Weather events such as flooding, protracted rains and intense cyclones are a result of climate change (Boschat et al., 2015; Easterling et al., 2000b; Meehl et al., 2000a; Nicholls and Alexander, 2007; Short and Neckles, 1999). Although a single weather event cannot be called climate change, there is mounting evidence that the weather patterns are changing as the concentration of greenhouse gases in the atmosphere continues to rise. The total amount of rainfall and average number of cyclones are not predicted to increase but there is an increase in frequency of intense rainfall events and severity of tropical cyclones. An increase in the frequency of extreme weather brings with it greater risks from floods, cyclones and higher water temperatures, increased frequency also shortens the time available for seagrass meadows to recover between events (Devlin et al., 2012a; Great Barrier Reef Marine Park Authority, 2011a; Gerald A Meehl et al., 2000; Gerald A. Meehl et al., 2000; Short and Neckles, 1999; reviewed in Wetz and Yoskowitz, 2013).

Due to climate change it is predicted that severe cyclones are going to occur more frequently, as the climate warms it brings a future where the recovery potential becomes increasingly important (Fuentes et al., 2012, 2011; Great Barrier Reef Marine Park Authority, 2011a; Hawkes et al., 2009, 2007).

1.7. Strandings

Strandings can occur for a variety of reasons including weather events, ingestion of synthetic materials, vessel strike, coastal development, tourism, increased incidence of disease, incidental catch in shark control program gear, and incidental capture in recreational and commercial fisheries gear (Caillouet Jr et al., 1991; Environment Australia, 2003; Flint et al., 2010b; Foley et al., 2012; Geraci and Lounsbury, 2005; Hazel et al., 2009; Hillestad et al., 1978; Limpus and Reed, 1985b; Marsh et al., 1986; Murphy and Hopkins-Murphy, 1989; National Research Council Committee on Sea Turtle Conservation, 1990; Renaud et al., 1991; Witherington and Ehrhart, 1989). The identification of impact frequency and magnitude is necessary to assess potential consequences of human activities when developing management measures (Dobbs 2001). However, human impacts have a greater effect near shore (Dobbs 2001; Hazel and Gyuris 2006; Hazel et al. 2009) potentially positively skewing prevalence of anthropogenic causes when assessing stranding data alone.

The long-term study of stranded animals can provide important information about potential trends for at-sea threats (anthropogenic and natural), diseases (Balazs, 1991; Herbst, 1994; Lloyd and Ross, 2015; Scherer et al., 2014), geographic ranges, seasonal distribution and life history (McFee et al., 2006; Scherer et al., 2014). Long term stranding monitoring programs assess the impact of implemented management actions (Scherer et al., 2014).

StrandNet is the Queensland Department of Environment and Heritage Protection (EHP) database which records dead, sick and injured threatened marine animals for the entire coast of Queensland. Records are received from members of the public, and employees of EHP, Queensland Department of Agriculture, Fisheries and Forestry (DAFF) and the Great

Barrier Reef Marine Park Authority (GBRMPA), verified, collated and stored in this central database.

Due to the fact that the monitoring of marine vertebrates including turtles at sea can be expensive, the use of strandings can be an effective ancillary tool to provide minimum counts of at sea mortality and threats (Peltier et al., 2012). This is why studies such as these are important to allow managers to be better understand strandings and the mechanisms behind them.

Norman et al.(2012), stated that the ability to understand and investigate marine mammal unusual mortality events and other unexpected strandings that involve substantial die-offs of the marine mammal population are important events which serve as indicators of ocean health, which can give larger insight into larger environmental issues, which may have implications for human health and animal welfare. Being able to understand the triggers and mechanisms causing strandings has important ramifications for ecosystem health.

The Queensland Department of Environment and Heritage Protection acknowledges numerous limitations to StrandNet including:

- The mass natural mortality occurring on Raine Island and Moulter Cay is not reported in this database, it is recorded in the turtle nesting database
- Animals caught and released as part of the Queensland Department of Agriculture, Fisheries and Forestry Shark Control Program are not reported in StrandNet
- There is less coverage of strandings in the sparsely populated areas of the Gulf of Carpentaria, Torres Strait and the eastern Cape York Peninsula
- Most traditional and indigenous hunting is not reported
- Fisheries by-catch of commercial fisheries may be incomplete (Meager and Limpus, 2012a)

During 2011, there was a higher proportion of strandings reported in the Gladstone & Townsville regions, with more than 3 times the total annual reports received for those regions (Meager and Limpus, 2012a). This has increase in strandings warranted further investigation, partly covered in this study and others including Flint et al., (2014); Gaus et al., (2012); Limpus et al., (2011).

It has been suggested that marine turtle stranding numbers follow seasonal trends influenced by weather events as well as land-based and at-sea seasonal activities. There have been links made between extreme weather and increased strandings (Flint et al. 2015; Marsh and Kwan 2008; Meager and Limpus 2012).

1.7.1. Background to the use of stranding data

There must be caution used when making management decisions based on stranding data alone, as the ecological significance of the examined stranding is often unknown. Some of the limitations of using stranding data are that the geographical origins of the animals are not known, and there is disputed credibility of the statistics related to strandings mainly because the sampling is opportunistic (Peltier et al., 2012).

In their paper Peltier et al.(2012) attempted to assess the quantitative significance of stranding events as an estimation of the fraction of cetacean carcasses that were drifting as opposed to those that washed ashore. Their aim was to improve the significance of cetacean stranding data by better understanding the drifting mechanisms of cetaceans at sea. With an understanding of the mechanisms such as currents, distance from coastline, atmospheric pressure, wind speed, carcass buoyancy and predation that a carcass experiences at sea, there is an ability to be able to locate the likely areas that the animal died (Epperly et al., 1996; Flint and Fowler, 1998; Leeney et al., 2008; McFee et al., 2006; Peltier et al., 2012). They found that 57% and 87% of stranded common dolphins originated from within the 100 m and 500 m isobars respectively (Peltier et al., 2012). This may contribute to determining the cause and location of the original incident/cause of stranding.

Epperly et al.(1996), analysed the use of stressed or dead turtles found on beaches as an index of at sea-mortalities. Between November 1991 and February 1992 Epperly et al., (1995) estimated that approximately 89-181 turtles were killed as a result of trawl fishing activities, however during subsequent analysis by Epperly et al.(1996) only 12 strandings were reported that could be related to these activities. There are many factors which bias this index, the most important of these is wind and ocean currents which determine the distance and direction that an animal can travel before stranding or whether the animal will wash ashore or not.

1.7.2. Numbers of turtles which strand

During 2011, in Queensland, there was a significant increase in the number of turtles with natural causes of death (45%) (Meager and Limpus, 2012b), compared to 1-7% during the 2005-2010 period (Biddle and Limpus, 2011). This increase has led to numerous investigations to examine live and dead stranded turtles and mammals, this in turn has led to an improved knowledge about mortality rates and causes, which has allowed a better understanding of population threats and stressors (Brodie et al., 2014; Flint et al., 2014, 2010d; Gaus et al., 2012; Limpus et al., 2011; Meager and Limpus, 2014; Norman et al., 2012). This understanding of strandings has also increased our ability to determine when a stranding situation may be 'unusual'. into the causes of increased strandings.

Improved understanding of the factors which cause marine turtles to strand will help management agencies to better manage these threats, thus contributing to the conservation of the species (Work et al., 2015).

For the purposes of management information, three scenarios have been identified to categorize factors that contribute to mortality in turtles: Human related (boat strike, fishing entanglement, legal hunting), natural (disease or congenital defect) and undiagnosed (usually where the carcass is too decomposed or unrecovered to allow a diagnosis). For green turtle deaths in 2011, 63% were undiagnosed, 16% were attributed to human related injuries, and 21% were natural causes. This contrasts the previous year's findings with 441 reported deaths, 72% of which were undiagnosed, 22% were attributed to human related injuries, and only 6% were due to natural causes. The most concentrated area of strandings occurred in the 28° to 25° latitudinal block (Gold Coast to Hervey Bay area)(41%, n= 728); followed by the 21° to 18° block (Mackay to Cardwell area)(30%, n=534) (Meager and Limpus, 2012a)(See Figure 3.1 for a map). It is acknowledged that some areas of the Gulf of Carpentaria, eastern Cape York Peninsula, Torres Strait are data deficient in terms of strandings information (Meager and Limpus, 2012a), it is unknown if there are any other areas of Eastern Queensland that are not monitored.

1.8. Rehabilitation

1.8.1. The need

Human activities in the ocean are continuing to cause the rapid depletion of marine megafauna worldwide, this in combination with the direct exploitation and unsustainable levels of incidental bycatch are the major threats facing turtles (Cardona et al., 2012; Lewison et al., 2004a; Moore et al., 2007). The stranding of injured animals is one symptom of these human interactions that attracts a lot of public interest. As a result of this increased public awareness, considerable resources are often allocated to the rehabilitation of stranded individuals (Cardona et al., 2012; Moore et al., 2007). For some, the justification for undertaking rehabilitation is to attempt to counter the effects of anthropogenic impacts (Mullineaux, 2014).

It is likely that extreme weather events, such as those seen in 2010 and 2011, negatively influence marine turtle population survivorship by increasing strandings. Rehabilitation of weather-related stranded turtles has been proposed to improve population survivorship by returning them to the ocean when their health has been restored and the environment has recovered from the impact. A large amount of resources (profit organization offsets, labour, infrastructure and public donations) are used annually to rehabilitate marine turtles. However, very little work has been done to determine the success of rehabilitation as a conservation strategy to help preserve endangered marine turtle populations (Baker et al., 2015; Feck and Hamann, 2013; Karesh, 1995; Moore et al., 2007; Tribe and Brown, 2000).

The rehabilitation of marine megafauna is driven by concern for the welfare of individual species (Cardona et al., 2012; Moore et al., 2007). Over recent years the number of animals being released to the wild with the alleged purpose of enhancing wild populations has increased as has the interest in them. The rehabilitation and release of wildlife as a conservation tool for the enhancement of populations has recently become more frequent (Baker et al., 2015; Cardona et al., 2012; Karesh, 1995; Mestre et al., 2014). Despite these increases the number of individuals being released into the wild is often too small to have any significant effect on these populations (Baker et al., 2015; Cardona et al., 2012; Feck and Hamann, 2013; Moore et al., 2007; Quakenbush et al., 2009).

Internationally there is interest in the need to treat wildlife and also educate the public but there are differing frameworks depicting the care of wildlife worldwide and domestically (Mullineaux, 2014). Among veterinarians, biologists and rehabilitation providers there is ethical debate regarding the treatment of animals. Some regard the only role of veterinarians as ending suffering through euthanasia (Loftin, 1985; Mullineaux, 2014) , while others promote the benefit of treatment and rehabilitation (Kirkwood, 2003; Mullineaux, 2014). The treatment of sick and injured wildlife creates a “feel good factor” for those involved (Cooper and Cooper, 2006; Kirkwood, 2003; Saran et al., 2011; Sikarskie, 1992) as well as providing educational benefit (Wobeser, 2007).Dubois (2003) found that the improvement of public awareness and education was found to be a close second priority to the provision of individual care, by people involved with rehabilitation.

Despite the need for increased public awareness, the welfare of individual animals must remain the top priority at all times of the rehabilitation processes, even over the personal and professional development that may occur as a results of prolonged care (Cooper and Cooper, 2006; Mullineaux, 2014).

1.8.2. Role of rehabilitation

Throughout Australia there are numerous marine turtle rehabilitation centres operating with the dual aims of contributing to the conservation of marine turtle populations and contributing to environmental education and public awareness (Feck and Hamann, 2013).However the magnitudes of these roles varies across the world (Mullineaux, 2014). Despite the costs involved, rehabilitation continues to be a tool for conservation as it provides a platform to educate members of the public about threats to marine turtle survival (Addison and Nelson, 2000; Feck and Hamann, 2013). It has been shown that when people visit zoos or aquariums that have a prominent conservation message then the visitors’ mindsets can be changed towards being more pro-conservation (Adelman et al., 2000; Falk et al., 2007; Wyles et al., 2013).

There is no question rehabilitation plays an important role in the care of individual animals to reduce suffering and treat certain conditions. In most cases the primary objective of wildlife rehabilitation is the welfare of individual animals (Baker et al., 2015; Moore et al., 2007; Saran et al., 2011), although sometimes the message is different in that rehabilitation is having a population conservation focus. Part of the rehabilitation process can include euthanasia as a treatment option to prevent individual suffering and can add value to research if appropriate post-mortem investigations are carried out. Even if animals cannot be successfully rehabilitated, a lot can be learned from the animal by conducting necropsies and post-mortem investigations. This has been evidenced in other species by the identification of novel pathogens not previously encountered in particular species/taxa (Barlow et al., 2012, 2010). However, with respect to contribution to the population of marine turtles, rehabilitation of individuals may not contribute to survivorship of the population. This is influenced by the size and health of the local turtle population, the factors affecting stranding and the conservation status of the local population.

1.8.3. Rehabilitation process

Rehabilitation is one of the most wildly used but poorly documented practices in wildlife conservation (Saran et al., 2011). In Australia, rehabilitation does not have standardised guidelines. Instead, each facility participating in marine animal care and rehabilitation is limited by their facility's mission and capacity as well as recommendations imposed by permitting in each local region (for example, local government ordinances and state government requirements). For example, the "Code of Practice – Care of Sick, Injured or Orphaned Animals in Queensland" (Department of Environment and Heritage Protection, 1992) is available for reference in Queensland but it is not a required protocol. Consequently, diagnostic procedures, treatment regimes, and duration in care vary between facilities and compared to other facilities internationally. This does not mean that welfare and animal care are not considered paramount. Several confounding factors apply in Australia with animals sent to rehabilitation based on field triage, accessibility of the animal to transport and resource availability to retrieve and receive the animal.

There is a lack of published information, particularly peer-review literature which relates to veterinary care and the treatment of wildlife (Mullineaux, 2014). This is particularly important considering that the standards of care and facilities vary enormously worldwide and domestically (Mullineaux, 2014).

One important concept in the welfare of individual animals is the 'triage' of animals to allow the quick euthanasia of animals that are unlikely able to be released into the wild (Mullineaux, 2014). During this triage process, there are several factors which should be considered, of which some are non-veterinary in nature which are likely to influence the success of treatment and rehabilitation (Mullineaux, 2014; Wobeser, 2007). Some of these factors include: - facilities available, suitably trained personnel, veterinary services available, funding, availability of release sites, significance of the individual, probability of success, consequences of no treatment and indirect benefits of treatment (Wobeser, 2007).

Rehabilitation will continue in some form for perpetuity as in most cases the public will not stand by and do nothing, despite the costs, success rate or population significance (Estes, 1998). Due to this questions still need to be asked about whether these efforts for individuals are keeping with the goals of conserving and protecting populations, species and ecosystems (Estes, 1998). The ideal goal of rehabilitation should encompass both the individuals and populations (Estes, 1998).

1.8.4. Success

It is difficult to assess the true success of rehabilitation without following each individual. In order to determine the success of rehabilitation, animals that are released need to be tracked (Grogan and Kelly, 2013).The two most appropriate methods for assessing post-rehabilitation survivorship are satellite-tracking individuals or tagging individuals and monitoring for their restranding or recapture with time. Queensland has provided an ideal opportunity for a case study of this issue due to the long running programs of both stranding and routine population monitoring.

There have been few studies investigating whether marine turtles are able to successfully readapt after rehabilitation, specifically with individuals that have required long and complicated veterinary treatment (Cardona et al., 2012; Feck and Hamann, 2013; Tomás et al., 2001). Information about the re-adaptation of rehabilitated marine turtles is scarce and has been restricted to looking at marine turtles which have been lightly incidentally captured by long-liners and released after on-board hook removal (Sasso and Epperly, 2007; Swimmer et al., 2006) or to individuals entangled in trammel nets and released a few hours later (Cardona et al., 2012; Snoddy and Southward Williard, 2010). In most cases rehabilitated animals are too elusive and conditions are not conducive to post release monitoring (Estes, 1998).

The fact that turtles have displayed abnormal behaviour post release from rehabilitation facilities casts doubt on the value of rehabilitation as a conservation tool (Cardona et al. 2012). Rescue and rehabilitation facilities definitely play an important role in public awareness and sample collection, although the goal of releasing animals for conservation purposes to offset human interactions needs to be investigated (Cardona et al., 2012).

Baker et al. (2015) conducted a study on the success of marine turtles in rehabilitation. They focused on age classes and outcome from rehabilitation. They found in Florida that 63% of turtles admitted to rehabilitation were not released back into the wild. They found most mortality occurred early in the processes but there were animals which died after long periods of care. They found some significant species and age-class survivorship. However, a major fault in this study was that they didn't account for cause of stranding or recapture post release. They openly acknowledge this and that future studies need to take this into account.

There have been numerous studies on head-started turtles after their release into the wild. This studies have shown mixed results. A study by Swingle et al., (1994) showed that even in the absence of injury or illness head-started turtle exhibited abnormal buoyancy patterns. In contrast the study by Polovina et al. (2006) compared captive-raised and wild loggerhead and didn't show any difference in dispersal patterns. Nichols et al. (2000) studied a wild-caught loggerhead which migrated back to Japan after it spent 10 years in captivity in Mexico. Head-started loggerhead turtles have provided some evidence that prolonged stays in captivity, even with the absence of illness or injuries can cause abnormal buoyancy (Addison and Nelson, 2000; Swingle et al., 1994) but there are different studies which showed that captive-raised and wild loggerheads didn't differ in their dispersal patterns (Cardona et al., 2012; Polovina et al., 2006).

Nichols et al.(2000) illustrated that wild caught loggerheads from the Japanese population that migrated back to Japan when they were released after spending 10 years in captivity in Mexico. Based on this study they suggested that a prolonged stay in captivity was unlikely to hamper the capacity of wild-born turtles to navigate and forage in the open ocean (Nichols et al., 2000).

The release of wild rehabilitated specimens has had only moderate success in the case of marine mammals and marine birds. Rehabilitated specimens have often displayed abnormal behaviour, dispersal patterns, reduced reproductive success and can experience low survival rates (Altwegg et al., 2008; Anderson et al., 1996; Bellido et al., 2010; Bettinger and Bettoli, 2002; Cardona et al., 2012; Ebner and Thiem, 2009; Estes, 1998; Fleming and Gross, 1993; Mazzoil et al., 2008; Mullineaux, 2014; Nawojchik et al., 2003; Nichols et al., 2000; Polovina et al., 2006; Thomas et al., 2010; Wells et al., 2009; Wolfaardt et al., 2009). Without the use of post-release monitoring the true success of rehabilitation cannot be assessed (Cooper and Cooper, 2006; Mullineaux, 2014).

Further, animals from numerous species often displayed abnormal behaviour, aberrant dispersal patterns, reduced reproductive success and experienced low survival rates post from care (Altwegg et al., 2008; Anderson et al., 1996; Bellido et al., 2010; Bettinger and Bettoli, 2002; Cardona et al., 2012; Ebner and Thiem, 2009; Fleming and Gross, 1993; Mazzoil et al., 2008; Nawojchik et al., 2003; Nichols et al., 2000; Polovina et al., 2006; Thomas et al., 2010; Wells et al., 2009; Wolfaardt et al., 2009).

Post-release studies can also help in regards to determining the best time and location to release animals from rehabilitation (Mullineaux, 2014). Numerous authors have found that releases are more likely to be successful when food is plentiful and any environmental stressors are reduced (Fajardo et al., 2000; Mullineaux, 2014; Tribe et al., 2005).As in the case reported by (Saran et al., 2011) Queensland marine turtle rehabilitation is driven by individuals or volunteer organizations. The success of such rehabilitation process is not fully understood. The determination of whether rehabilitation has been successful or not should not be determined based on individual treatment but in combination with the individuals long term survival (Saran et al., 2011)

There is nothing known about the capacity of released animals to start breeding again and contribute to population maintenance after rehabilitation (Baker et al., 2015; Cardona et al., 2012; Karesh, 1995; Mestre et al., 2014). Tomás et al.(2001) found during their study that the full recovery and survivorship of loggerhead turtles after release from fishing hook interactions was possible if only for a short time. The long term survivorship success of turtles after rehabilitated as demonstrated by successful reproduction remains unclear, with many exhibiting behavioural anomalies while they were tracked (Cardona et al., 2012).

Therefore, given the resources used in rehabilitating marine turtles, assessing the capacity of these species to readapt to the wild, including their ability to survive and reproduce, are essential to guarantee that resources allocated are maximising the number of marine turtles contributing to the functional population (Baker et al., 2015; Cardona et al., 2012; Flint et al., 2015). Queensland has provided an ideal opportunity for a case study of this issue due to the long running programs of rehabilitation, stranding and routine population monitoring.

1.8.5. Facilities

While the exact costs of rehabilitation are not published, the cost of treating a marine turtle is thought to vary considerably between centres and individuals, and is assumed to be high given the time, facility, staff and finance commitments that are needed to care for individuals (Feck and Hamann, 2013).

Funding for rehabilitation comes from public donations, philanthropic trusts or corporations, some also receive regular or periodic government funding and/or charge entrance fees to observe animals (Feck and Hamann, 2013).

Despite the costs involved, rehabilitation continues to be an important tool for conservation as it provides a platform to educate members of the public about threats to marine turtles survival (Feck and Hamann, 2013). Rehabilitation centres save individuals which otherwise would have died, but they also play a larger role in the education and public awareness, this role is not well documented (Feck and Hamann, 2013).

Rehabilitation centres educate visitors about the causes of marine turtle injuries and illnesses and the actions they can take in their everyday lives to help to conserve them and the environment in which they live (Feck and Hamann, 2013) .

Studies shown that learning about conservation together with observing wildlife up close is more effective at changing the attitudes of visitors towards the conservation of marine turtles (Ballantyne et al., 2007; Feck and Hamann, 2013; Tisdell and Wilson, 2005). Seeing injured wildlife close up is speculated to stimulate visitors to either change their behaviour and/or donate money for their conservation. This has been seen with turtles at nesting beaches in studies done by Ballantyne et al., (2007); Tisdell and Wilson, (2002;

2001). Tisdell and Wilson (2003) and Wilson and Tisdell (1999), suggested that this additional revenue can be put towards further research into threat mitigation and management, they also suggest that the tourism economic vale can contribute to the legal support for threat mitigation, such as the enforcement of ‘go slow zones’ and marine littering. These previous studies have focused on the marine turtle nesting beaches and have not focused on the economic and educational value of rehabilitation facilities. Feck and Hamann (2013), surveyed visitors to four rehabilitation facilities throughout Queensland and found that all visitors had learnt about the threats to marine turtles and that they were willing to undertake at least one change in their everyday lives to help minimize these threats to marine turtles.

Rehabilitation and rescue centres play a major role in environmental education, public awareness and sample collection. All of these tasks are valuable and should not be abandoned and the release of rehabilitated turtles should continue. They are all particularly useful awareness activities (Cardona et al., 2012).

1.9. Statistics

The use of modelling to explore the relationships between response (or dependent) variables and explanatory (or independent) variables is an important tool in economics and social sciences (Burnham and Anderson, 2002; Gasparrini, 2011; Graham, 2003; Zeileis et al., 2008). When this relationship displays a delayed effect complications arise, which require the development of more complex models (Gasparrini, 2011). This delay is termed a lag and it defines the time interval between the exposure and the outcome (Gasparrini, 2011).

The objective of time series analysis is the development of mathematical models that provide plausible descriptions for sample data, in terms of variation in the dependent variable being explained by the independent variable (Bhaskaran et al., 2013; Shumway and Stoffer, 2011). General Linear Models can be used to model count time series, by using lagged values of dependent variables to account for autocorrelation (McLeod et al., 2011).

The challenge of identifying factors associated with an ecological phenomenon is one often faced by ecologists, conservation biologists and wildlife managers (Murray and

Conner, 2009). In an ideal world scientists would be able to manipulate all variables, however in many cases this is not an option due to financial, logistical or ethical reasons (Graham, 2003; Murray and Conner, 2009). This necessitates the need to conduct multivariate analysis to identify the “best” models or suite of models (Murray and Conner, 2009).

While constructing models the goal should be to build a model which explains the greatest variability in the response variable with the fewest number of explanatory variables (Graham, 2003).

Most count data have excess numbers of zeros, which is called *zero inflated* (Zeileis et al., 2008; Zuur et al., 2009). Histograms or frequency plots can be used to detect zero inflated data sets (Zuur et al., 2009). There are several methods which can be used to deal with these including zero-inflated Poisson and zero-inflated negative binomial models (Zuur et al., 2009), these methods are briefly discussed as follows. If zero-inflation is ignored there is a risk the estimated parameters and standard errors could be biased or the excessive number of zeros could cause overdispersion (Zuur et al., 2009). The use of Poisson or negative binomial dispersions are still options when using zero-inflated methods (Chandler and Andrew Royle, 2013; Kéry and Schaub, 2012; Martin et al., 2015; Zeileis et al., 2008; Zuur et al., 2009).

Classical modelling methods involving Poisson regression models are often of limited use in ecology as count data is typically over-dispersed or has an excess number of zeros (Bhaskaran et al., 2013; Zeileis et al., 2008; Zuur et al., 2009). There are numerous other methods to use in order to capture the over-dispersion, including quasi-Poisson, negative binomial, which all belong the family of generalised linear models (Zeileis et al., 2008). GLMs all use the basic log-linear mean function for the model, they describe the dependence of on variable on another or others (Zeileis et al., 2008).

The following points were considered when choosing modelling methodology. Quasi-Poisson deals with overdispersion by using the mean regression function from a standard Poisson model but leaves the dispersion parameter to be determined by the data as opposed to being set at 1 (Zeileis et al., 2008). Negative binomial arise as a gamma mixture of Poisson distribution (Zeileis et al., 2008). An advantage of undertaking the negative binomial regression is that a formal likelihood is associated with it, thus producing a readily available information criteria (Zeileis et al., 2008).

The first attempt in the investigation of relationships between variables is to complete the basic Poisson regression models. This will help to highlight any potential issues with overdispersion (Zeileis et al., 2008). The next steps is completing the quasi-Poisson regression and then moving onto the more formal method to deal with over-dispersion negative-binomial regression (Zeileis et al., 2008).

The method in which the best model is selected is the focus of most statistical analysis (Burnham and Anderson, 2004, 2002; Doherty et al., 2012; Link and Barker, 2006; Murray and Conner, 2009).Before model selection can occur models must first be constructed, they be constructed either a priori or in combination with model selection activities such as those in stepwise procedures (Doherty et al., 2012)

Burnham and Anderson (2002); and Doherty et al. (2012) both support the fact that analysts start with a scientific hypothesis and develop a set of concise models (4-20). They also recognise that in some circumstances there be closer to 100 models however Anderson (2008) stipulate that the number of models should not exceed the sample size (Doherty et al., 2012). It is advised the method of examining “all possible models” practically those which focus on stepwise selection process should be avoided as they often produce spurious results (Burnham and Anderson, 2002).

Murray and Conner (2009) point out that just because correlation is found between variables this does not indicate causation. Murray and Conner (2009) propose that model analysis is becoming more important/prominent as resource management agencies are faced with shrinking budgets as it can assist them in prioritizing management strategies and allocating resources to their most productive use. Modelling can be used to develop testable hypotheses in regards to the creation of alternative management strategies (Murray and Conner, 2009).

Model selection can be attained using multiple methods, one of which is the use of AIC values (Burnham and Anderson, 2002). When using quasi distributions a modified AIC can be calculated based on quasi-likelihoods (Burnham and Anderson, 2002).

AIC (Akaike's Information Criterion) is based on the relationship between the relative expected Kullback-Leibler distance and the maximized log-likelihood (Burnham and Anderson, 2002). The Kullback-Leibler distance can best be theorised as a directed distance between two models (Burnham and Anderson, 2002). AIC is calculated using the equation $AIC = -2\log(\mathcal{L}\hat{\theta}|y)) + 2K$. When using quasi-distribution for overdispersed data the equation becomes $QAIC = -\left[2\log(\mathcal{L}(\hat{\theta}))/\hat{c}\right] + 2K$. AIC and QAIC are on a relative scale and are strongly dependent on sample size (Burnham and Anderson, 2002). When analysing AIC it is not the absolute size of the value which is important it is the relative value to other models AIC which is important (Burnham and Anderson, 2002). There are several ways in which to measure and access this value including AIC differences $\Delta i = AIC_i - AIC_{min}$ and "Akaike weights" $w_i = \frac{exp\left(-\frac{1}{2}\Delta_i\right)}{\sum_{r=1}^{R}\exp\left(-\frac{1}{2}\Delta_r\right)}$. Both of the aforementioned methods allow for models to be ranked and therefore scientific hypothesis to be tested (Burnham and Anderson, 2002).

AIC can only be used to compare models created with the same data sets (Burnham and Anderson, 2002).

Burnham and Anderson (2002) advocate the use of published literature and experience in the biological sciences in the formation of a priori candidate models. This often includes the creation of a global model which has numerous parameters based on the "science of the situation" and reflects the study design and attributes of the system studied (Burnham and Anderson, 2002). From this point alternative models with fewer variables can be constructed to represent plausible alternatives. These alternative models generally involve different numbers of parameters.

Burnham and Anderson (2002) strongly advocate the exclusion of variables which do not make biological sense. They also advise the inclusion of all models that have reasonable justification prior to analysis. After the completion of the running of the models, the analyst has the task of interpreting the evidence left from the data (Burnham and Anderson, 2002). From this some questions, can be answered objectively, allowing for consideration of past studies, biological information.

Distributed lag models is a method used to analyse the delayed effects between the exposure and the outcome (Gasparrini, 2011). They allow the effects of an exposure to be distributed over a period of time (Gasparrini, 2011). In order to perform distributed lag non-linear models, all methods require the transformation of the original predictor variable in order to generate new variables to be used in the model (Gasparrini, 2011).

1.10. Conclusions

When looking at the literature reviewed in this document, there is numerous information about the rehabilitation success of other species but not marine turtles. It has also been demonstrated that there is a lack of knowledge about the effects that extreme weather events have on marine turtles and how weather/environmental variables can be used to predict marine turtle stranding rates.

Chapter 2. Objectives & Hypothesis

2.1. Objectives

Through collaboration between the School of Veterinary Science, the Queensland Department of Environment and Heritage Protection and private organisations (Sea World, Australia Zoo and Underwater World – SEA LIFE), this study was conducted to advance the understanding of marine turtle strandings caused by extreme weather events and the survivorship of these turtles after rehabilitation, with these specific aims:

1) Examine the current literature to understand current state of knowledge for marine turtle stranding trends and the links between extreme weather events.
2) Examine the strandings between 1996 and 2013 to get a better understanding of the trends and cycles
3) Examine the causes of animals being sent to rehabilitation and the outcome for each specific one
4) Examine the link between stranded turtles and their input into key wild populations
5) Help management agencies to designate appropriate rehabilitation facilities.
6) Increase the level of understanding about the implications that extreme weather events cause to marine turtles
7) Allow the prediction of stranding rates following extreme weather events to allow facilities to better respond to increases in stranded animals
8) Through the increased understanding of their effects, determine the net benefit of rehabilitation and better predict and prepare for the effects of future extreme weather events.
9) Develop methods for consideration by management agencies to recognize and respond to mass/unusual mortality or disease events appropriately.

2.2. General hypothesis

These aforementioned aims cumulatively tested the null hypothesis that:

“*Rehabilitation is a viable practice used to successfully treat and return green and loggerhead marine turtles to their resident grounds after a catastrophic event.*”

2.3. Structure of the thesis

To meet the aims of the study, this thesis is a retrospective data analysis used to test trends and create predictive models of specific catastrophic events.

Chapter 1 reviewed the relevant literature to identify the deficits in our current state of knowledge of marine turtle strandings to address **Objective 1**. It became apparent that robust baseline data for marine turtle strandings in Queensland were not available. This baseline data required for further analysis (**Chapter 3**) were analysed to address **Objective 2**. Investigations continued (**Chapter 4**) into the cause of marine turtles being sent to rehabilitation and the outcomes of these animals, this was used to address **Objective 3, 4 and 5**. Comparisons were then made between animals sent to rehabilitation and animals who were triaged onsite and returned to the ocean, further addressing **Objective 4 and 5.** The trends identified in the preceding two chapters were used to model the relationship between marine turtle strandings and environmental factors (**Chapter 5**), this was to address **Objective 6**. Predictive modelling was then used to enable management agencies and rehabilitation facilities to be better prepared for increased numbers of marine turtle strandings (**Chapter 7**). This addressed **Objectives 7 and 8**. Overall findings are discussed in **Chapter 8** with and a concluding statement. The output produced throughout the thesis and the findings and concluding statement addressed **Objective 9**.

A **bibliography** completes the thesis and includes all articles used in all chapters.

Chapter 3 has been published as part of this investigation. **Chapter 4** has been accepted for publication as part of this investigation. **Chapters 5** and **6** have been submitted for publication as part of this investigation. The styles and reference styles have been altered to fulfil the requirements for The University of Queensland thesis.

All chapters including the bibliography, followed the reference format of *The Veterinary Journal*.

Chapter 3. Trends in Marine Turtle Strandings along the east Queensland, Australia Coast between 1996 and 2013.

This chapter is the first of its kind as a preliminary trend analysis for Queensland marine turtle strandings numbers. The dataset used in this analysis is the same dataset used throughout the whole thesis. This paper provided insight into the trends and cycles of marine turtle strandings in Queensland between 1996 and 2013, and provided a launching pad for the remainder of the thesis.

Published as:

Flint J, Flint M, Limpus CJ, Mills PC. Trends in Marine Turtle Strandings along the East Queensland, Australia Coast, between 1996 and 2013. J Mar Biol. 2015;2015. doi:10.1155/2015/848923

3.1. Abstract

In-water monitoring of marine vertebrates is usually expensive while the use of stranding data can be used to provide a cost-effective estimation of disease and mortality. Strandings for Queensland are recorded in a web based database (StrandNet) managed by the Queensland Government's Department of Environment and Heritage Protection (EHP). Data recorded in StrandNet from the east coast of Queensland between 1996 and 2013 were investigated for patterns of stranding. Significant trends in Queensland over this time were: (i) An increase in the number of animals reported stranded within this study site; (ii) a species (loggerhead and green marine turtles) prevalence; (iii) a seasonal effect on different age classes stranding with most overall strandings occurring between August and November; and (iv) stranding hotspots (Moreton Bay, Hervey Bay, Rockhampton region and Cleveland Bays) persisting throughout the study timeframe. This study suggested that intervention strategies such as rehabilitation, should be able to be focussed on periods of heightened importance and specific localities to minimise health risks and contribute to sustainable use of resources.

3.2. Introduction

All six species of marine turtles found within Australian waters are listed as species of conservation concern under the *Environmental Protection and Biodiversity Conservation Act 1999* and in Queensland waters under the Queensland *Nature Conservation Act 1992*. Marine turtles are protected within a series of marine parks along the coastline as prescribed under the *Marine Parks Act 2004 (Qld)* and the *Great Barrier Reef Marine Park Act 1975 (Commonwealth)*. Monitoring stranded marine turtles along the Queensland coast provides a measure of the effectiveness of these legislations and other temporary protection measures.

The monitoring of marine vertebrates, particularly marine turtles, in water can be expensive. Peltier et al. (2012) assessed the quantitative significance of stranding events as an estimation of the fraction of cetacean carcasses that were drifting as opposed to those that washed ashore. They found that 57% and 87% of stranded common dolphins originated from within the 100 m and 500 m isobaths, respectively (Peltier et al., 2012). This suggested that stranding data may be used to identify trends and potential issues occurring in the near shore environment but inferences to at-sea deaths cannot be drawn.

Strandings can occur for a variety of reasons including ingestion of synthetic materials, vessel strike, coastal development, tourism, increased incidence of disease, incidental catch in shark control program gear, and incidental capture in recreational, commercial fisheries gear and unknown reasons (Environment Australia, 2003; Flint et al., 2010d; Hazel et al., 2009; National Research Council Committee on Sea Turtle Conservation, 1990). The identification of impact frequency and magnitude is necessary to assess potential consequences of human activities when developing management measures (Dobbs, 2001). However, human impacts have a greater effect near shore (Dobbs, 2001; Hazel et al., 2009; Hazel and Gyuris, 2006) potentially positively skewing prevalence of anthropogenic causes when assessing stranding data alone.

It has been suggested that marine turtle stranding numbers follow seasonal trends influenced by weather events as well as land-based and at-sea seasonal activities. There have been links made between extreme weather and increased strandings (Flint et al., 2014; Marsh and Kwan, 2008; Meager and Limpus, 2012a).

This study investigated 18 years of marine turtle stranding data along the Queensland coast, compiled using the StrandNet database. The overall trend of strandings, sex, age class and species distributions for season and known environmental impacts at selected locales were examined to interrogate the database for any variances in stranding that may elucidate factors involved in stranding events.

3.3. Methods

13854 turtles were reported stranded between 1996 and 2013 along the eastern Queensland coast. For each turtle a minimum of age, sex, species, fate of carcass, location, time and cause of stranding was recorded.

3.3.1. Data

StrandNet is the Queensland Government's Department of Environment and Heritage Protection (EHP) state-wide database which records dead, sick and injured threatened marine animals for the entire coast of Queensland and adjacent Commonwealth waters. Records are received from members of the public, and employees of EHP, Queensland Parks and Wildlife (QPWS), Queensland Department of Agriculture, and Fisheries (DAFF) and the Great Barrier Reef Marine Park Authority (GBRMPA). Information is collated and stored in this central database. Once reports are entered by on-ground staff the information available is verified by regional and state co-ordinators for standardisation.

3.3.1.1. *Biometrics (Age, sex, species)*

Standard measurements such as curved carapace length (CCL) and tail to carapace length (TCL) were collected (Limpus et al., 1994a).

Sex was determined by gonad examination by trained personnel either onsite or using photographs or measurements (Limpus and Limpus, 2003; Limpus and Reed, 1985a).

Species was determined as one of six turtle species including subspecies (green *Chelonia mydas*, loggerhead *Caretta caretta*, flatback *Natator depressus*, hawksbill *Eretmochelys imbricata*, leatherback *Dermachelys coriacea*, olive ridley *Lepidochelys olivacea*, black turtle *Chelonia mydas agassizi*), as a hybrid animal or species unknown based on dichotomous key characteristics (Environmental Protection Agency, 2008; Great Barrier Reef Marine Park Authority, 2007).

3.3.1.2. *Location*

Study area encompassed latitude -10.78° to -28.16° and longitude 142.15° to 155° **(Figure 3.1)**. The east coast of Queensland was selected as it has a long term and complete dataset; with data collection biased to regions of survey and higher populations. This limitation is openly acknowledged by Meager & Limpus (2012) but considered valid as a representative of a minimum recovery rate and indicative of trends occurring.

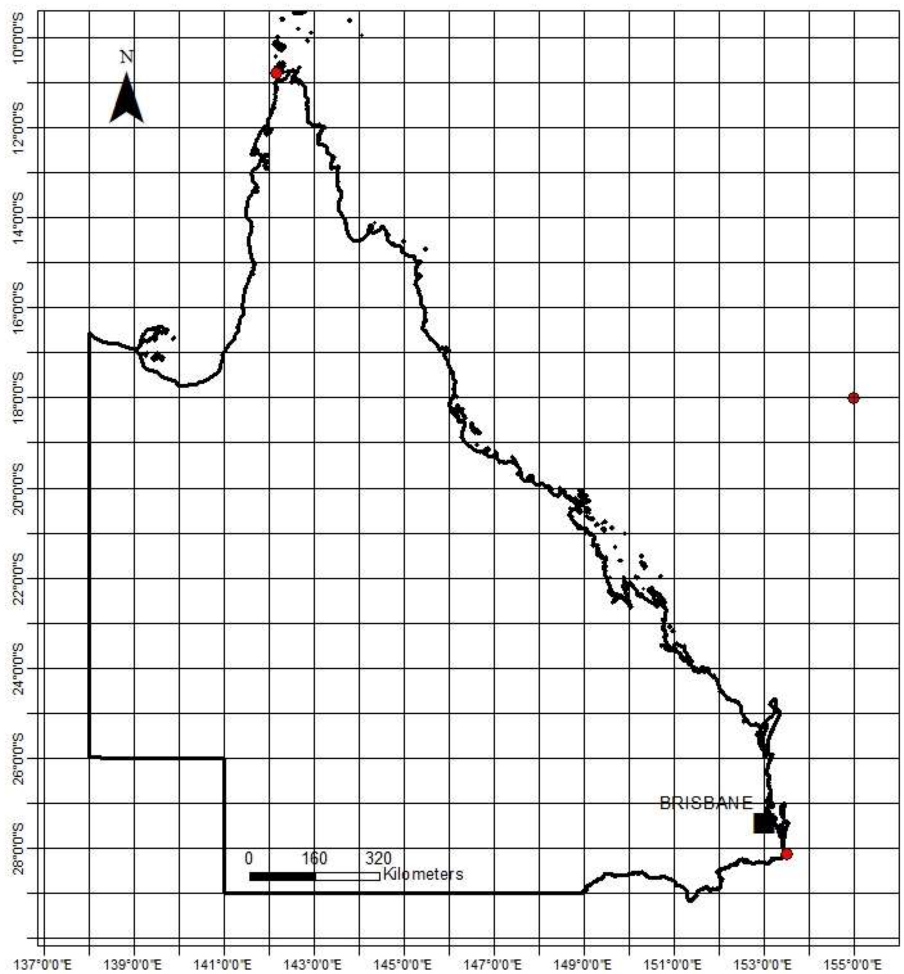


Figure 3.1 Map of Queensland coast showing the extents for which data was used. Red dots denote limits to study area.

3.3.1.3. *Time*

The date a turtle was reported stranded was used as a proxy of time of death providing month and season: Summer (December to February), Autumn (March to May), Winter (June to August) and Spring (September to November).

3.3.1.4. *Cause of Stranding*

The term 'stranding' is used here to incorporate all reported sick, injured, incapacitated or dead marine turtles that were either found ashore or, in rare cases, were encountered at sea. It includes turtles which were entangled in fishing nets, synthetic debris or rescued from a situation where they would have died had it not been for human intervention (such as being found on their back on a nesting beach or being flipped on back due to falling or surf) (Geraci and Lounsbury, 2005).

Within StrandNet, the primary cause of death/stranding was identified based on gross examination, photograph and/or necropsy by trained personnel. Cause of stranding identified in StrandNet was based on the summation of information available.

3.3.2. Statistical analyses:

3.3.2.1. *Biometrics (Age, Sex, Species)*

Animals were pooled into four age classes, as follows: small immature, large immature, adult sized and unknown. Age class is only an approximation of maturity. It does not confirm reproductive development. The breakdown of age class for loggerheads were adapted from Limpus et al., (1994b), hawksbills from Limpus, (1992) and other species were adapted from Limpus et al., (1994a).

Animals were pooled based on gender as males, females and unknowns. Unless an internal gonadal examination was conducted, animals were sexed based on TCL and CCL measurements. Sex determination for larger animals was based on the ratio of these two measurements (Limpus, 2007, 1992, Limpus et al., 1994a, 1994b; Limpus and Limpus, 2003) .

Gender analysis did not exhibit sexual dimorphism for any age class, so subsequent analyses for sex were pooled.

3.3.2.2. *Location*

The latitude recorded in StrandNet was used to map the occurrence of strandings along the coast to identify the distribution and highlight potential “hot spots”. As the exact location where a stranding was reported was not necessarily where the impact/incident occurred, strandings were grouped into latitudinal blocks of 0.5° to account for this potential error.

3.3.2.3. *Time*

Boxplots were used to illustrate the number of turtles stranded per month across all years. This was done to illustrate potential seasonal trends.

Rate of strandings throughout the year were compared using chi-squared tests to determine variance between expected and observed rates for each species. Expected rates were defined to be equal distribution throughout the year for each group analysed.

The same test was applied to evaluate if there was a difference between the age classes of each species. It was expected that the total number of strandings would be evenly distributed throughout the year. Expected values were rounded up to the nearest whole number. All statistical analysis was performed using R (R Core Team, 2016).

In order to assess the seasonality of trends, the series was broken down into its three components, using the "decompose ()" function in R: trend, seasonal effect and randomness. The series was seasonally adjusted by subtracting the estimated seasonal component from the original data. This data was then plotted to show the trend and the irregular components (Coghlan, 2014).

Autocorrelation function techniques were used to visually display potential seasonal patterns with the data.

3.3.2.4. *Causes of stranding and mortality*

The identified causes of "mortality" were grouped into 6 categories: unknown, natural, release, rehabilitation, anthropogenic and depredation. Descriptive statistics were used to compare between season, year, age and sex.

3.4. Results

A total of 13 854 marine turtle strandings records from 1996-2013 were examined.

3.4.1. Biometrics (Age, Sex, Species)

Total number of strandings for each species and age class showed that the observed number was significantly different to the expected numbers of loggerhead small immatures, loggerhead adult sized, loggerhead large immatures, green large immatures, and green adult sized, green small immatures and unknown species (**Table 3.1)**.

Table 3.1 Chi-squared total strandings by species and age class between years (18 years or data)

Species and age class	χ^2	df	p
Loggerhead small immatures	64.47	17	< 0.001
Loggerhead adult sized	53.33	17	< 0.001
Loggerhead large immatures	217.22	17	< 0.001
Green large immatures	254.31	17	< 0.001
Green adult sized	514.29	17	< 0.001
Green small immatures	2535.92	17	< 0.001
Turtle small immatures	705.36	17	< 0.001
Turtle large immatures	116.93	17	< 0.001
Turtle adult sized	481.22	17	< 0.001
Hawksbill Small immatures	227.21	17	< 0.001

More small immature greens and unknown species were observed while fewer large immature greens and loggerhead turtles stranded when comparing 1996 to 2013 **(Table 3.2).**

Table 3.2 Distribution between age classes

Species	Age class	1996	2013	R^2
Green turtles	small immature	21.8% (n=69)	56.1% (n=494)	0.7525
	large immature	32.3% (n=102)	14.8% (n=128)	0.6975
	adult sized	35.1% (n=111)	25.7% (n=226)	0.5147
	unknown	10.8% (n=34)	3.6% (n=32)	0.0329
Loggerhead turtles	small immature	9.6% (n=8)	23.3% (n=7)	0.2874
	large immature	48.2% (n=40)	30% (n=9)	0.5856
	adult sized	31.3% (n=26)	46.7% (n=14)	0.166
	unknown	10.84% (n=9)	0% (n=0)	<0.001
Unidentified turtles	small immature	11.25% (n=13)	23.08% (n=49)	0.6132
	large immature	6.25% (n=5)	4.62% (n=9)	0.0922
	Adult sized	16.25% (n=13)	25.13% (n=49)	0.0127
	unknown	66.25% (n=53)	47.18% (n=92)	0.1865

The most commonly reported stranded marine turtle species was green (69.6%, 9641/13854, 95% CI 0.69 - 0.70), loggerhead (7.8%, 1081/13854, 95% CI 0.07 - 0.08), hawksbill (5.9%, 813/13854, 95% CI 0.05-0.06) and then others (flatback, ridley, hybrids, black and leatherback; 1.5%, 201/13854, 95% CI 0.01 - 0.02). In addition, unidentified turtle species accounted for 15.3% (2118/13854, 95% CI 0.15-0.16).

3.4.2. Location

The majority of strandings occurred in the -27.0, -23.5, -19.0 latitudes, corresponding with coastal big cities and catchment outflows. Latitudes outside of these hotspots showed that there were peaks in different latitudes during different years. These peaks were of a smaller magnitude and not consistent.

3.4.3. Time

The number of strandings over time from 1996 to 2013 showed seasonal variation with peaks in October and troughs in March–June **(Figure 3.2 and 3.3).** Examination of data for green strandings shows that different age classes had different timing for peaks of strandings. Adults and large immature turtle strandings peaked in October while small immatures turtle strandings peaked in August. The observed number of strandings for this species varied significantly throughout the year for all years with the exception of 2000 ($\chi^2 = 17.79$, df=11, $\rho = 0.0868$). Loggerhead turtles showed some variance to this pattern with two cycles annually.

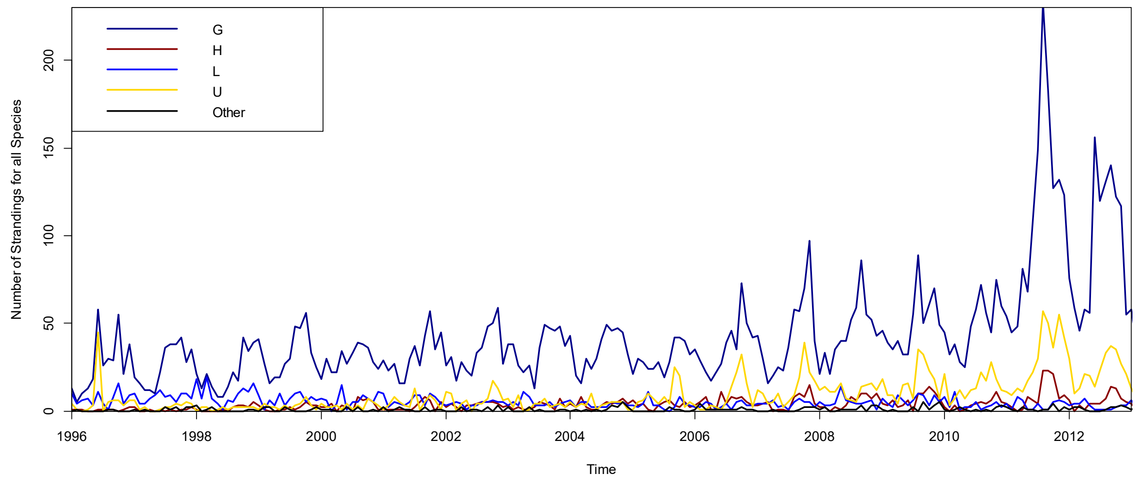


Figure 3.2. Total number of marine turtle strandings reported to StrandNet on the eastern Queensland coast for each species.

G= *Chelonia mydas*, H= *Eretmochelys imbricata*, L =*Caretta caretta*, U= unidentified turtle, Other = *Chelonia mydas agassizi*, *Dermachelys coriacea*, *Natator depressus*, *Lepidochelys olivacea* and *Caretta caretta* x *Chelonia mydas* hybrid.

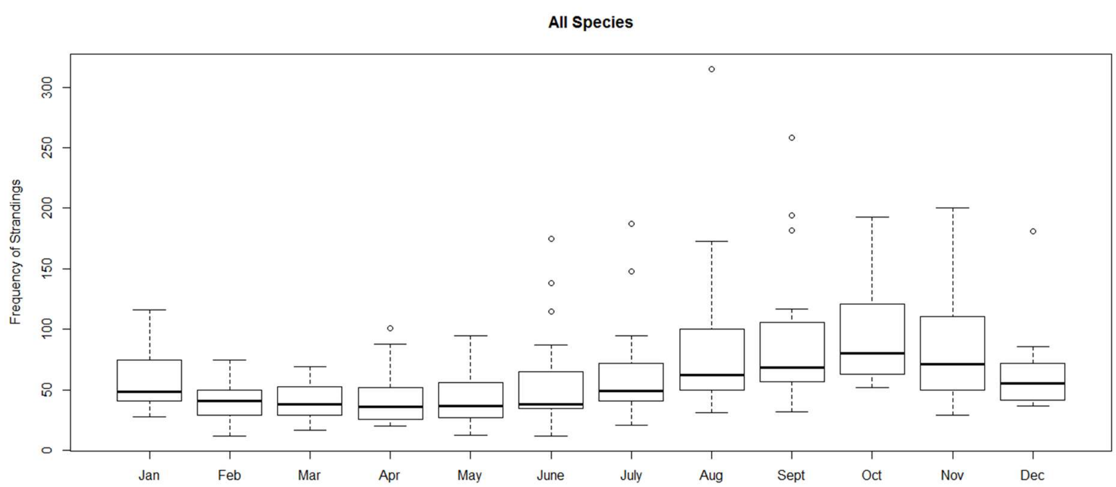


Figure 3.3 Box plot of total monthly stranding values.

Observed (1st plot) stranding has a general upward trend (2nd plot) and a strong seasonal component (3rd plot) **(Figure 3.4)**

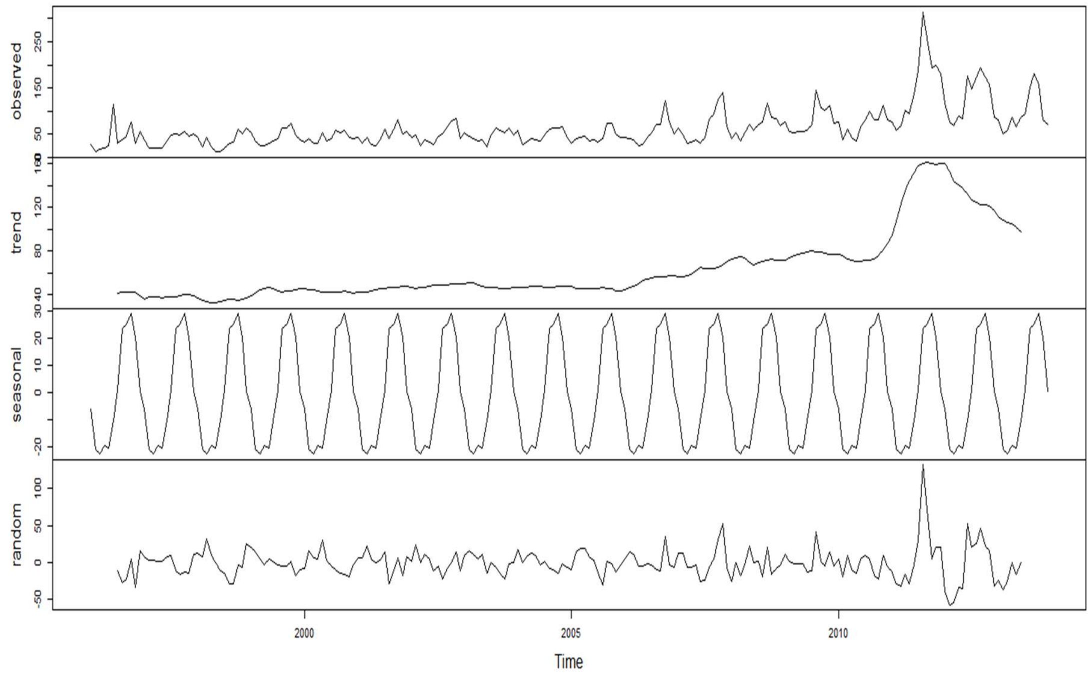


Figure 3.4. Decomposition of additive time series of total monthly strandings.

For all strandings the largest seasonal factor was October (28.82) and the lowest was March (-22.36), indicating peak in strandings in Spring and a trough in strandings in Autumn each year **(Figure 3.4)**.

Autocorrelation techniques support the significant strong annual cycle to marine turtle strandings at the state level seen in **Figure 3.4**.

3.4.4. Cause

Natural causes contributing to mortality has varied since 1996. The proportion of anthropogenic and unknown causes of death has declined. The proportion of depredated animals and animals released onsite has remained consistent. The number of animals sent to rehabilitation has increased over the years **(Figure 3.5).**

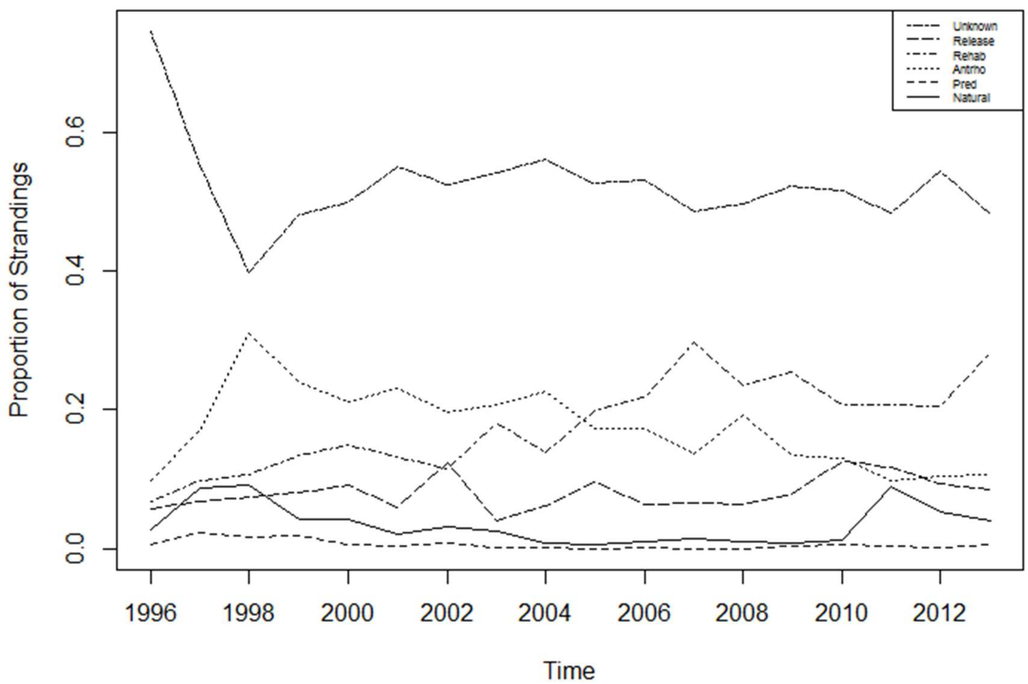


Figure 3.5. Proportion of turtles with an identified outcome.

3.4.5. Species

The number of green turtles which have been reported stranded has increased from 1996 to 2011 but subsequently decreased **(Figure 3.2)**. The proportion reported has remained consistent, with a small increase, ranging between 60 and 75% (R^2= 0.0949).

The number of loggerhead turtles which have been reported stranded has fluctuated since 1996 **(Figure 3.2)**. There has been a decrease in the proportion of loggerheads that have stranded since 1996 (R^2=0.7609).

The observed number of monthly turtle strandings between years showed a significant difference to the expected numbers of green strandings (χ^2 = 624.82, df=187, ρ < 0.001), loggerhead strandings (χ^2 = 278.72, df=187, ρ < 0.001), hawksbill (χ^2 = 228.39, df=187, ρ < 0.001) and unidentified turtles (χ^2 = 742.62, df=187, ρ < 0.001).

Total number of observed strandings between years showed a significant difference to the expected numbers of green turtles (χ^2 = 2789.45, df=17, ρ < 0.001), hawksbill (χ^2 = 233.85, df=17, ρ < 0.001), loggerhead (χ^2 = 156.43, df=17, ρ < 0.001), unidentified turtles (χ^2 = 1258.35, df=17, ρ < 0.001).

3.5. Discussion

Overall this study found temporal, spatial and age related patterns in the numbers of marine turtle strandings. Given these recurrent patterns, further investigation is warranted to develop models that predict the resultant increases in the numbers of stranding from each of these confounding factors to determine when to mitigate negative impacts.

This study shows years of elevated strandings for all age classes in marine turtles in general, and specifically all age classes of green turtles and loggerhead turtles **(Table 3.1**).

Between 1996 and 2013 the most frequent species recorded as stranded were greens and loggerheads (*n*=10722, 77%) **(Figure 3.2**); and of the 13854 turtles reported stranded on Queensland coastline, there was a prevalence of dead green turtles, irrespective of age class (69.6%). Both of these species are common residents of Queensland waters, whereas the olive ridley, black and leatherback have relatively lower population numbers within these waters (Environment Australia, 2003; Limpus, 2009, 2008a, 2008b, 2008c, 2007).

The increase in the numbers of juvenile green turtles which strand over the 18-year study could be due to several issues including the increase or development of emerging age specific impediments, and increase in the size of the population. This could be influenced by small immature turtles being immunologically naïve and susceptible to environmental stressors. New diseases or the coastal and catchment urbanization and climate change that Queensland is experiencing may be impacting this least robust cohort of the population (Flint et al., 2014, 2010c, 2010d).

When pooled for age classes there is a visible cyclical trend of strandings occurring through the year for all turtles, greens, hawksbills and unknown species. This uneven distribution throughout the year indicates that there may be underlying confounding processes linked to season that is influencing the rate of stranding. Time series analysis **(Figure 3.2 and 3.3)** showed that turtle stranding is cyclical across years with more turtles stranding during the months coming out of winter (August to November) and fewer turtles stranding in the months when waters start to cool (April to June). Further periods of unusual extreme weather may result in outliers in this normal seasonal patterns (Flint et al., 2014; Marsh and Kwan, 2008; Meager and Limpus, 2014, 2012a). These outliers warrant independent investigation as they relate to periods of increased need for resources and rehabilitation if turtle deaths are to be minimized by intervention.

Strandings were distributed along the Queensland coast in localised “hotspots”. These hotspots correspond to the semi-enclosed embayments of Moreton Bay, Hervey Bay, Rockhampton region and Cleveland Bay. The hot spots are also in the vicinity of major catchments areas along the coast including the Brisbane, Fitzroy, Burnett, Burdekin Rivers. This is important because it highlights where extra resources are required and brings in local areas which warrant further investigation.

The number of animals without an identified cause of death has remained at a high level since 1996. This could be due to the condition of the carcasses when they are found, inexperienced observers or a lack of funds/resources to conduct adequate analysis. The identification of causes of mortality is an essential step involved in the understanding the health of individuals and the long term health of the population (Flint et al., 2009) and in turn, can be used as a sentinel of environmental health (Aguirre and Lutz, 2004) and management priorities.

Through the years there has been an increase in the number of animals which have been sent to rehabilitation centres **(Figure 3.5)**. This has correlated with a shown need and resultant increase in the number of centres which provide care. Despite this there has been no study conducted into the proportion of these animals which are released and survive or subsequently re-strand. This of particular interest to know the overall benefit of rehabilitation.

Anthropologic causes of death have decreased over the years which supports the hypothesis that current management actions such as go slow zones, TEDs, protection areas and net attendance rules are successful as mitigation strategies **(Figure 3.5)**. The other identified causes of stranding have remained at a low level.

Even though the number of dead turtles that strand is only an index on the actual number of animals which die in total (Epperly et al., 1996; Peltier et al., 2012), monitoring stranding of marine turtles along the coastline provides a powerful first line tool in gathering data to make management decisions. It is now imperative this data be used to advance other tools such as modelling to accurately predict important habitats, patterns, needs, and resource allocation to mitigate marine turtle deaths. As marine turtles are facing the same threats globally, this strategy could be implemented elsewhere and used as a uniform step-wise approach to objectively assess coastline and rehabilitation centre management. Once implemented, success needs to be measured over medium to long-term (10 year) trends and be treated as a dynamic plan that is adjusted as any issues are identified.

This study showed the lowest stranding rates occurred in the large immature population of marine turtles in Queensland but all of the population is influenced by annual seasonal effects with stranding rates being exacerbated by extreme events.

The authors declare that they have no conflict of interest.

This article does not contain any studies with animals performed by any of the authors.

This study was partially funded by the Australian Research Council Linkage Project Grant LP110100569.

We would like to thank the Queensland Marine Wildlife Strandings and Mortality network and all contributors to the StrandNet database. We would also like to thank turtle rehabilitation facilities staff at Underwater World-SEA LIFE (Mooloolaba, QLD), Australia Zoo (Beerwah, QLD and SeaWorld (Gold Coast, QLD) for providing access to their data.

Chapter 4. Status of Marine Turtle Rehabilitation in Queensland.

This chapter analyses the data recorded within StrandNet of animals that were stranded alive and sent to rehabilitation facilities in Queensland. Analysis was then conducted to determine the success of rehabilitation as a conservation tool for marine turtle populations. There has been limited analysis of survival of animals post release from rehabilitation, with exception of some satellite tracking. This provides the first study combining StrandNet data and the Queensland Turtle Conservation Project data. This study was used as a case study to provide rehabilitation facilities with information about prioritising resources to ensure the best outcome for turtles to contribute to a functioning wild population.

This article has been accepted in PeerJ.

Flint J, Flint M, Limpus CJ, Mills PC.2017 Status of Marine Turtle Rehabilitation in Queensland. PeerJ. 2017;Accepted.

4.1. Abstract

Rehabilitation of marine turtles in Queensland has multifaceted objectives. It treats individual animals, serves to educate the public, and contributes to conservation. We examined the outcome from rehabilitation, time in rehabilitation, and subsequent recapture and restranding rates of stranded marine turtles between 1996 and 2013 to determine if the benefits associated with this practice are cost-effective as a conservation tool.

Of 13854 marine turtles reported as stranded during this 18-year period, 5022 of these turtles stranded alive with the remainder verified as dead or of unknown condition. A total of 2970 (59%) of these live strandings were transported to a rehabilitation facility.

Overall, 1173/2970 (39%) turtles were released over 18 years, 101 of which were recaptured: 77 reported as restrandings (20 dead, 13 alive subsequently died, 11 alive subsequently euthanized, 33 alive) and 24 recaptured during normal marine turtle population monitoring or fishing activities.

Of the turtles admitted to rehabilitation exhibiting signs of disease, 88% of them died, either unassisted or by euthanasia and 66% of turtles admitted for unknown causes of stranding died either unassisted or by euthanasia. All turtles recorded as having a buoyancy disorder with no other presenting problem or disorder recorded, were released alive.

In Queensland, rehabilitation costs approximately $1,000 per animal per year admitted to a centre, $2,583 per animal per year released, and $123,750 per animal per year for marine turtles which are presumably successfully returned to the functional population. This practice may not be economically viable in its present configuration, but may be more cost effective as a mobile response unit. Further there is benefit in giving individual turtles a chance at survival and educating the public in the perils facing marine turtles. As well, rehabilitation can provide insight into the diseases and environmental stressors causing stranding, arming researchers with information to mitigate negative impacts.

4.2. Introduction

The nearshore waters of Queensland, Australia, provide important marine turtle nesting and foraging grounds that support a significant proportion of the South Pacific Ocean loggerhead (*Caretta caretta*) genetic stock and the southern and northern Great Barrier Reef green turtle (*Chelonia mydas*) populations (Dobbs, 2001; Limpus and Reimer, 1990; Slater et al., 1998).

Throughout Australia there are numerous marine turtle rehabilitation centres operating with the aim of contributing to the conservation of marine turtle populations (Feck and Hamann, 2013). In recent years rehabilitation centres have played a dual role: (i) saving individuals which may have otherwise died if they had not received medical attention; and (ii) contributing to environmental education and public awareness (Feck and Hamann, 2013); with the former having a two-fold benefit of keeping individuals alive and conservation of the species.

In Australia, rehabilitation does not have national standardised guidelines. Instead, each facility participating in marine animal care and rehabilitation is limited by their facility's mission and capacity as well as by recommendations imposed by permitting in each region (for example, local government ordinances and state government requirements). For example, the "Code of Practice – Care of Sick, Injured or Orphaned Animals in Queensland" (Department of Environment and Heritage Protection, 1992) is available for reference in Queensland but it is not a required protocol. Consequently, diagnostic procedures, treatment regimes, and duration in care vary among facilities and when compared to other facilities internationally. This does not mean that welfare and animal care are not considered paramount. Several confounding factors apply in Australia with turtles sent to rehabilitation based on field triage, accessibility of the animal to transport and resource availability to retrieve and receive the animal.

As general public awareness for wildlife conservation has increased, there has been a corresponding increase in the numbers of stranded turtles reported, rehabilitated and subsequently released back into the wild with the intent of enhancing wild populations (Feck and Hamann, 2013). Although the rehabilitation of marine megafauna is driven by concern for the welfare of individual animals, the number of rehabilitated individuals may be too small to have any significant effect on the population or species (Baker et al., 2015; Cardona et al., 2012; Feck and Hamann, 2013; Moore et al., 2007; Quakenbush et al., 2009). Further, animals from numerous species often displayed abnormal behaviour, aberrant dispersal patterns, reduced reproductive success and experienced low survival rates post rehabilitation (Altwegg et al., 2008; Anderson et al., 1996; Bellido et al., 2010;

Bettinger and Bettoli, 2002; Cardona et al., 2012; Ebner and Thiem, 2009; Fleming and Gross, 1993; Mazzoil et al., 2008; Nawojchik et al., 2003; Nichols et al., 2000; Polovina et al., 2006; Thomas et al., 2010; Wells et al., 2009; Wolfaardt et al., 2009). As well, a large amount of resources (profit organization offsets, labour, infrastructure and public donations) are used annually to rehabilitate marine turtles in Australia but the benefit this is having on marine turtle populations remains unquantified.

Therefore, given the resources used in rehabilitating marine turtles, assessing the capacity of these species to readapt to the wild, including their ability to survive and reproduce, are essential to guarantee that resources allocated are maximising the number of marine turtles contributing to the functional population (Baker et al., 2015; Cardona et al., 2012; Flint et al., 2015). Queensland has provided an ideal opportunity for a case study of this issue because of the long running programs of routine population monitoring, stranding response and rehabilitation.

We investigated whether different causes of stranding as well as the length of time an animal spends in care influenced the long term survival of individuals during and post rehabilitation. We summarised and analysed the available data to provide rehabilitation facilities with options to undertake this method of species conservation.

4.3. Methods

4.3.1. Data

Data used in this study were obtained through StrandNet, the Queensland Government's Department of Environment and Heritage Protection (EHP) state-wide database reporting threatened stranded marine turtles for the entire coast of Queensland and adjacent Commonwealth waters as outlined in Flint et al. (2015). In brief, records were received from members of the public, and employees of EHP, Queensland Parks and Wildlife Services (QPWS), Queensland Department of Agriculture, and Fisheries (DAF), rehabilitation/triage centres (including but not limited to ReefHQ, Cairns Turtle Hospital, James Cook University, Quion Island Turtle Rehabilitation Centre, SeaWorld, Australia Zoo Wildlife Hospital and Underwater World-SEA LIFE Aquarium) and the Great Barrier Reef Marine Park Authority (GBRMPA). Information was collated and stored in this central database. Once reports are entered by first responders the information available is verified by regional and state coordinators for standardization. When required, a second opinion on cause of stranding is sought from experts such as wildlife veterinarians and senior environmental scientists. (For more information, see https://www.ehp.qld.gov.au/wildlife/caring-for-wildlife/strandnet-reports.html).

Additional data were obtained from the EHP Queensland Turtle Conservation Project (QTCP), SeaWorld, Australia Zoo Wildlife Hospital and Underwater World-SEA LIFE Aquarium. The data provided by SeaWorld, Australia Zoo Wildlife Hospital and Underwater World-SEA LIFE Aquarium were used to complete data in StrandNet such as outcomes, causes of death and duration of care.

The QTCP database is the Queensland Government's EHP state-wide database which records tagging and tag recaptures for all marine turtles encountered for the entire coast of Queensland and adjacent Commonwealth waters. Records are received from members of the public, trained volunteers and employees of EHP, QPWS, DAF and GBRMPA. Additional tag recoveries are reported by members of the public. Amalgamating these databases produced the first comprehensive dataset of strandings, causes and captures throughout Queensland for 1996 to 2013.

4.3.2. Categories used for data Analysis

Biometrics were assessed using standard measurements, gonad examination and/or dichotomous key characteristics. Age classes were broken down into three broad categories: small immature, large immature and adult-sized, based on curved carapace measurements, adapted from Limpus (1992) and Limpus, Couper & Read (1994a, 1994b).

Cause of stranding was identified by examining information compiled from first-responders and trained staff who reviewed reports, photos and codes recorded in StrandNet (Flint et al., 2015). All determinations of the cause of stranding were made within the StrandNet reporting mechanisms and verified outside of this study. The cause identified in StrandNet was then used to group causes for this analysis. Turtles often presented with multiple disorders but are recorded in StrandNet as the suspected primary cause of stranding or most obvious condition. For example, an animal presenting with a disease state causing a buoyancy disorder may only be recorded as "disease"; or an animal admitted floating with a fracture is recorded as fracture because it cannot be determined if it was floating before or only after the time of impact.

4.3.2.1. *Terms used throughout this study*

The term *stranding* is used to incorporate all reported sick, injured, incapacitated or dead marine turtles that either were found washed ashore or encountered at sea. It includes turtles which were entangled in synthetic debris including fishing nets and line, as well as turtles which were rescued (Biddle and Limpus, 2011; Flint et al., 2015; Geraci and Lounsbury, 2005; Meager and Limpus, 2012b). For each animal, a single primary cause of stranding was recorded.

Entanglement is defined as being entrapped in an anthropogenic object such as fishing line/rope/net.

Fracture is used to denote any form of fracture to a turtle that is attributable to anthropogenic causes (e.g. boat strike or blunt force trauma).

Disease is classified as turtles which exhibit protracted ill health from a cause consistent with a physiological condition and not otherwise caused by anthropogenic activity (e.g. fracture, entanglement). This is often linked with poor body condition.

Buoyancy disorder is used to describe turtles which were observed floating and in which no other presenting problem or disorders were observed.

Unknown cause of death/stranding was used when a cause could not be accurately determined. In most cases for cause of death this was due to there being no necropsy performed and no obvious external cause identifiable with a gross examination.

For this study, *survivorship* is defined as a turtle being found in good condition at least once after release from rehabilitation. Determination of condition was made based on coming onto a nesting beach or laparoscopic examination of the gonads or via in-water population surveys and being found to be in good condition at each capture.

Rehabilitation was deemed non-successful if the animal was reported stranded again (either dead or alive) within the timeframe of the collected dataset (1996-2013).

Duration in care was calculated by subtracting the date of outcome, from the original date of admission/stranding.

Rodeo is a technique developed to capture turtles during in-water surveys. This technique is presented pictorially in Limpus (1978) and described in depth in Limpus (1985). Briefly this technique involves the searching for turtles by traversing predefined sampling areas via boats, once observed a jumper dives on the turtle in order to catch and bring on board the vessel. Turtles were only captured during daylight hours. If more than one turtle was seen at the same time, the first to be sighted was pursued.

4.3.3. Data Analysis

Turtles were included if they stranded along the east Queensland coast within the area of latitude -10.78° (Cape York) to -28.16° (Queensland-New South Wales border) and longitude 142.15 to 155° (Flint et al., 2015). Recaptured turtles were included regardless of where they were encountered (e.g. overseas or in New South Wales) if their original stranding was within the defined Queensland coast.

Animals were matched between the databases using unique identifiers, such as titanium tag numbers. To find subsequent recaptures of the same animal, queries were performed with the capture date that was greater than the first recorded stranded.

Outcomes were analysed two ways: using the actual calculated time until outcome and then grouping time in care into three groups: admission to 7 days in care (short term stay), 7 to 28 days in care (medium term stay) and greater than 28 days in care (long term stay).

R was used to perform all statistical analysis described above (R Core Team, 2016). Results were presented as descriptive statistics with rigor expressed as a standard 95% Confidence Interval. Confidence intervals were selected as they represent the variance within a dynamic population.

When analysing time period between release and recapture numbers are expressed as averages, ± a standard deviation.

4.3.4. Rehabilitation costs

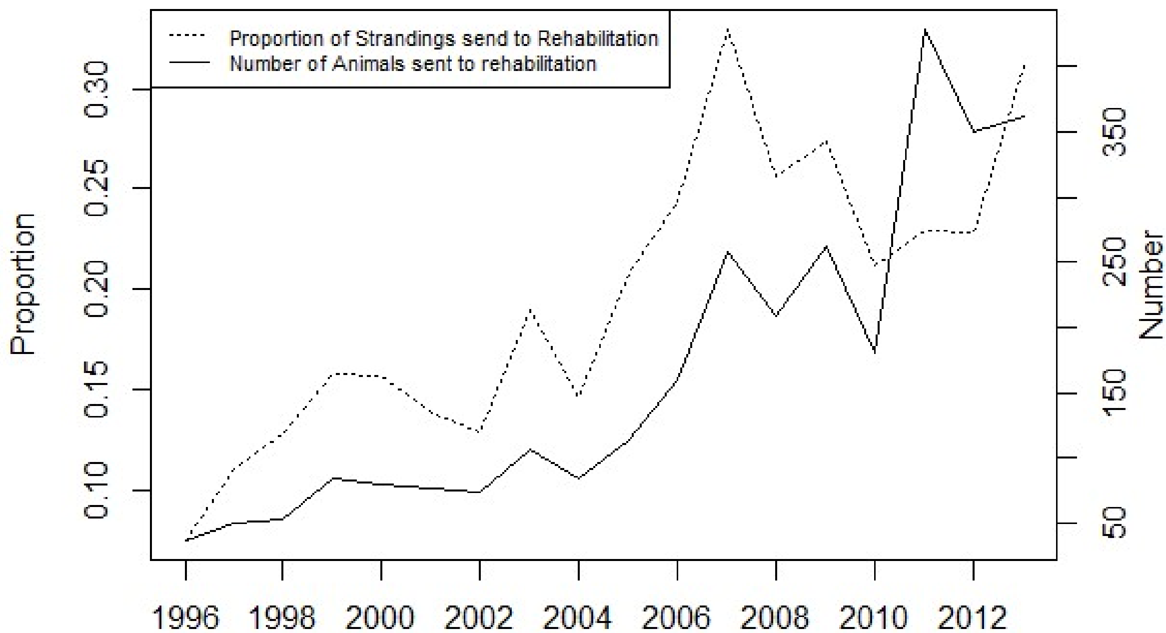
The three main rehabilitation facilities in Queensland were approached and asked to provide their estimated annual costs for marine turtle rehabilitation. These costs are derived based on pool maintenance, food, labour and some medical costs. They do not identify general operating costs that are absorbed by the facility. At best, they should be viewed as an estimate.

The three facilities rehabilitated approximately 68% of turtles admitted to rehabilitation in Queensland. The costs supplied were then extrapolated out to account for all rehabilitated turtles in Queensland, at best these costs should be viewed as a low estimate. Calculations were then made based on numbers of turtles admitted to rehabilitation, numbers of animals released from rehabilitation, numbers of turtles encountered again, and numbers of unsuccessful attempts at rehabilitation.

4.4. Results

4.4.1. Stranded animals sent to rehabilitation

Of the 13854 marine turtle strandings along the Queensland coastline between 1996 and 2013, 5022 of these animals stranded alive, of which 2970 (59.1%, 2970/5022, 95%CI 57.7-60.4%) were admitted to a rehabilitation facility. There was an increase over time in the number (R^2=0.70) and proportion (R^2=0.80) of stranded turtles which were sent to rehabilitation facilities in Queensland **(Figure 4.1**).



.

Figure 4.1. Animals sent to Rehabilitation in Queensland, n=1173.

Two thousand and twenty-one (68%, 2021/2970 rehabilitated turtles) were treated at three institutions: SeaWorld received 596 (20% of turtles sent to rehabilitation, 596/2970, 95%CI 18.6-21.5%), Australia Zoo Wildlife Hospital received 788 (26.5% of turtles sent to rehabilitation, 788/2970, 95%CI 24.9-28.1%), and Underwater World received 637 (21.4% of turtles sent to rehabilitation, 637/2970, 95%CI 20-22.9%).

4.4.1.1. *Species of stranded turtles sent to rehabilitation*

Green turtles were most often sent to rehabilitation both by number and by proportion (78.2%, 2324/2970, 95%CI 76.7-79.7%)**.** This was followed by hawksbill turtles, *Eretmochelys imbricata* (11.2%, 334/2970, 95%CI 10.1-12.4%) and loggerhead turtles (7.3%, 217/2970, 95%CI 6.4-8.3%) with the other species (flatback turtles, *Natator depressus* (1.2%, 37/2970, 95%CI 0.09-1.7%), olive ridley turtles, *Lepidochelys olivacea* (0.5%, 16/2970, 95%CI 0.3-0.9%), unknown (1.3%, 41/2970, 95%CI 1-1.8), black turtle, *Chelonia mydas agassizi* (0.03%, 1/2970, 95%CI 0.006-0.2)) remaining at low levels.

4.4.1.2. *Age class of stranded turtles sent to rehabilitation*

Consistently, over the years, the majority of turtles sent to rehabilitation were small immature sized turtles (71%, 2108/2970, 95%CI 69.3-72.6%)**.** The numbers of large immature (15%, 453/2970, 95%CI 14-16.6%) and adult sized (12%, 343/2970, 95%CI 10.4-12.7%) turtles admitted varied each year.

4.4.1.3. *Cause of stranding of stranded turtles sent to rehabilitation*

The most common cause of stranding for animals sent to rehabilitation was unknown (54%, 1613/2970, 95%CI 52.5-56.1%). The most common identified presenting problems or disorders were disease (18%, 530/2970, 95%CI 16.5-19.2%), buoyancy disorder (13%, 393/2970, 95%CI 12.1-14.5%) and fracture (6%, 167/2970, 95%CI 4.8-6.5%). **Figure 4.2** shows the proportions of animals sent to rehabilitation with identified causes.

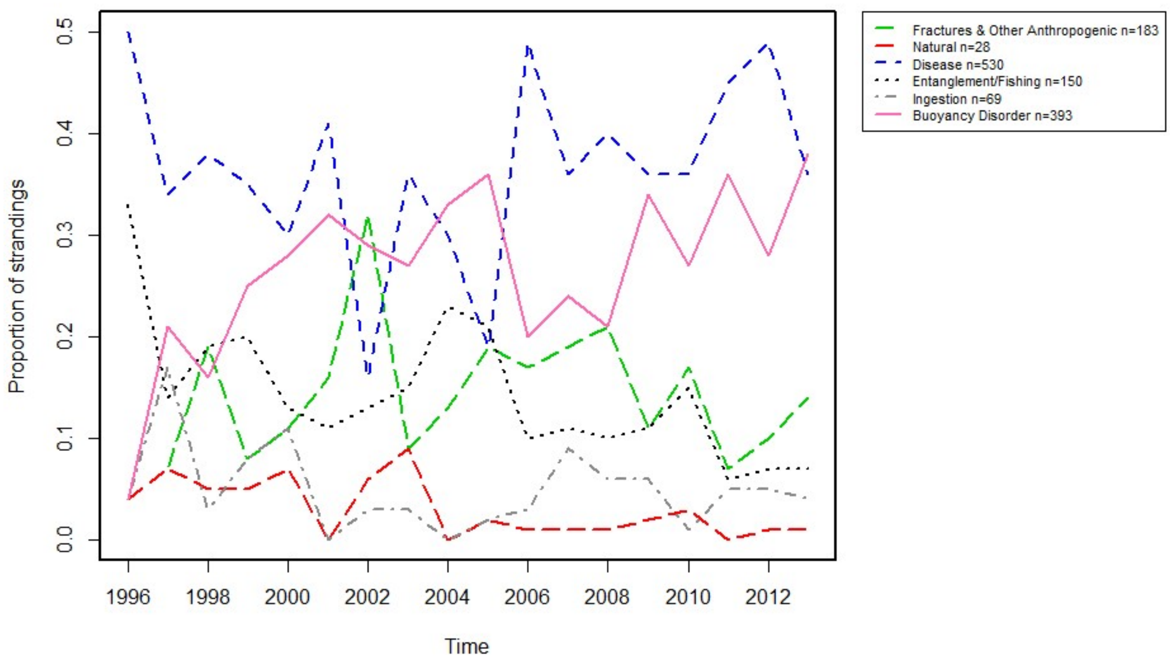


Figure 4.2. Proportion of animals sent to rehabilitation with an identified cause of stranding

4.4.1.3.1. *Age class*

Irrespective of age class, the most common record of stranding for turtles sent to rehabilitation by proportion was from unknown, followed by disease and buoyancy disorder (**Table 4.1**).

Table 4.1. Cause of stranding for turtles going to rehabilitation by age-class. Recorded as percentages per ageclass.

	Adult Sized	Small Immatures	Large Immatures
Boat Strike/Fractures	8.75	4.65	8.17
Buoyancy Disorder	11.95	13.71	13.47
Depredation	2.33	0.33	1.1
Disease	21.57	17.41	18.1
Dredging	0	0.05	0
Entangled Ghost fishing	0	0.09	0
Entanglement Crabbing	6.41	0.43	0.66
Entanglement fishing	2.04	2.37	1.99
Entanglement rope	0.58	0.38	0
Hunting	0.58	0	0.88
Ingestion of foreign material	3.21	1.85	3.97
Netting	0	0.24	0
Other Anthropogenic	0.29	0.43	1.32
Unknown	41.11	57.69	47.90
Unknown Natural	0.00	0.28	0.44
SCP	0.87	0.09	1.55
Nesting Beach	0.29	0	0
Land Reclamation	0	0	0.44

4.4.1.3.2. *Species*

For all species with the exception of the black and flatback turtles, the most common cause of turtles being sent to rehabilitation was for unknown reasons followed by disease then buoyancy disorders **(Table 4.2)**. For loggerhead turtles, fractures had the same proportion as buoyancy disorders. For flatback turtles, the most common cause was unknown, followed by buoyancy disorder then disease, and the only black turtle was admitted with ingestion of foreign material.

Table 4.2. Proportion of animals sent to rehabilitation by species and cause of stranding. Reported as percentages per species.

Cause of stranding	**Green turtle** (*Chelonia mydas*)	**Loggerhead turtle,** (*Caretta caretta*)	**Hawksbill turtle,** (*Eretmochelys imbricata)*	**Olive ridley turtle,** (*Lepidochelys olivacea)*	**Flatback turtle,** (*Natator depressus)*	**Black turtle** *(Chelonia mydas agassizi)*
Buoyancy Disorder	13.77	6.45	11.68	18.75	40.54	0
Fractures	6.2	6.45	2.1	0	2.7	0
Depredation	0.56	2.3	0	6.25	2.7	0
Disease	18.33	7.83	23.35	25	10.81	0
Dredging	0.04	0	0	0	0	0
Entangled Ghost fishing	0.09	0	0	0	0	0
Entanglement Crabbing	1.2	2.3	0	0	0	0
Entanglement fishing	2.24	2.3	2.4	0	2.7	0
Entanglement rope	0.39	0	0.9	0	0	0
Hunting	0.26	0	0	0	0	0
Ingestion of foreign material	2.62	1.38	0.9	6.25	0	100
Netting	0.22	0	0	0	0	0
Other Anthropogenic	0.56	0.92	0.3	0	0	0

Table 4.2 Continued

SCP	0.3	3.69	0.3	0	0	0
Unknown	52.97	64.98	58.08	43.75	35.14	0
Unknown Natural	0.17	0.92	0	0	5.41	0
Nesting Beach Rescues	0	0.46	0	0	0	0
Land Reclamation	0.09	0	0	0	0	0

4.4.2. Fate of turtles sent to rehabilitation

Between 1996 and 2013, there were changes in the outcomes of turtles sent to rehabilitation **(Figure 4.3).** Overall between 1996 and 2013, the proportion of turtles euthanized increased (R^2=0.47), while the proportion of turtles which died while in care decreased (R^2= 0.38). Between 1996 and 2013, the proportion of turtles which were released was highly variable with a slight decrease (R^2=0.07).

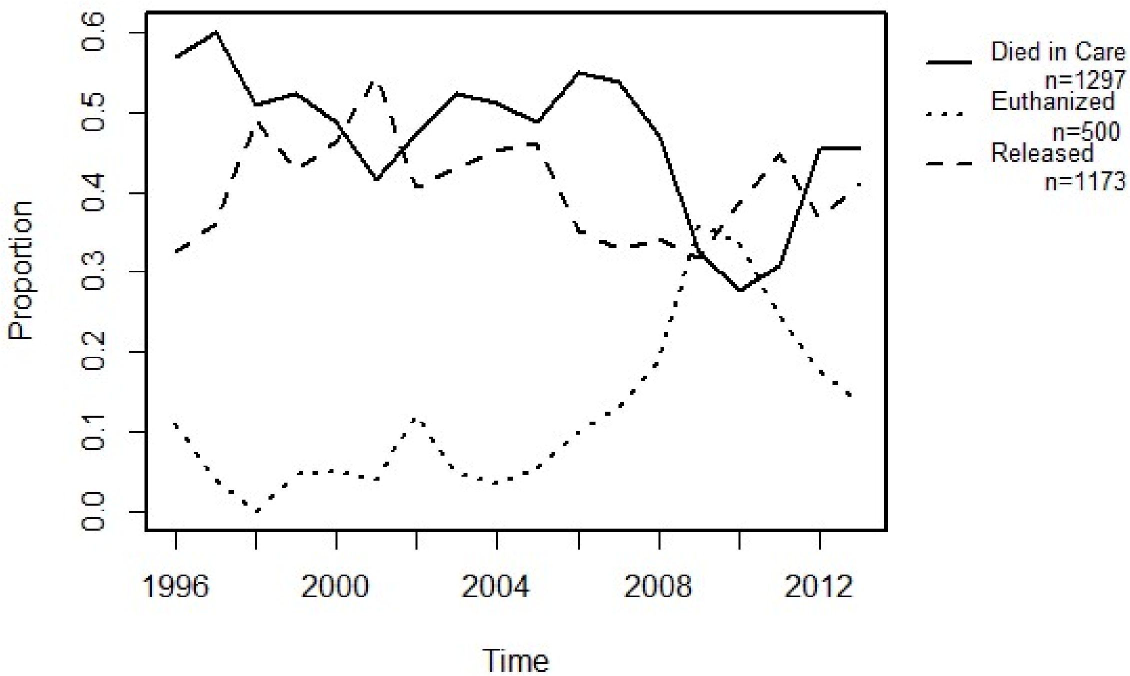


Figure 4.3. Proportions of animals per outcome.

Between 2005 and 2009, there was a steady increase in the proportion of turtles euthanized which reversed after this time. During this same period there was an inversely proportional decrease in the number of unassisted deaths in care **(Figure 4.3).** Of the turtles which were sent to rehabilitation 39% (1173/2970, 95%CI 37.7-41.2%) were released (**Table 4.3**)

Table 4.3. Cause of stranding distributed by outcome of rehabilitation. The numbers by outcome, animals and cause are reported as percentages

Cause of stranding	Died in Care				Euthanized				Released			
	n	% by outcome	% of all animals	% by cause	n	% by outcome	% of all animals	% by cause	n	% by outcome	% of all animals	% by cause
Buoyancy disorder	0	0	0	0	0	0	0	0	393	33.50	13.23	100
Fractures	63	4.8	2.12	37.72	62	12.40	2.09	37.13	42	3.58	1.41	25.15
Depredation	9	0.6	0.30	45	2	0.40	0.07	10	9	0.77	0.30	45
Disease	309	23.8	10.40	58.30	157	31.40	5.29	29.62	64	5.46	2.15	12.08
Dredging	1	0.08	0.03	100	0	0	0	0	0	0	0	0
Entangled ghost fishing	0	0	0	0	1	0.20	0.03	25	3	0.26	0.10	75
Entanglement crabbing	2	0.15	0.07	5.41	2	0.40	0.07	5.41	33	2.81	1.11	89.19
Entanglement fishing	17	1.31	0.57	24.64	12	2.40	0.40	17.39	40	3.41	1.35	57.97
Entanglement rope	5	0.39	0.17	38.46	2	0.40	0.07	15.38	6	0.51	0.20	46.15
Hunting	0	0	0	0	0	0	0	0	6	0.51	0.20	100

Table 4.3 Continued

Ingestion of foreign material	49	3.78	1.65	71.01	19	3.80	0.64	27.54	1	0.09	0.03	1.45
Netting	0	0	0	0	1	0.20	0.03	20.00	4	0.34	0.13	80
Other anthropogenic	13	0	0.44	81.25	2	0.40	0.07	12.50	1	0.09	0.03	6.25
SCP	5	0.39	0.17	31.25	1	0.20	0.03	6.25	10	0.85	0.34	62.50
Unknown	818	63.07	27.54	50.71	239	47.80	8.05	14.82	556	47.4	18.72	34.47
Unknown natural	3	0.23	0.10	75	0	0	0	0	1	0.09	0.03	25
Nesting beach rescues	3	0.23	0.10	60	0	0	0	0	2	0.17	0.07	40
Land reclamation	0	0	0	0	0	0	0	0	2	0.17	0.07	100
Grand Total	1297	100	43.67	NA	500	100	16.84	NA	1173	100	39.49%	NA

When combining the proportion of animals which died while in care (assisted and unassisted) and comparing against the number of animals which were released, the proportion of animals which died slightly increased over time (R^2=0.07), while the proportion of animals which were released decreased over time (R^2=0.07). Both patterns showed a lot of variability.

4.4.2.1. *Cause of Stranding for euthanized, died unassisted and released rehabilitated turtles*

Table 4.3 shows the outcomes for all turtles sent to rehabilitation (n=2970). Overall, 500 rehabilitated turtles were euthanized. The cause of stranding for the highest number euthanized were unknown (48.7%, 239/500, 95%CI 43.4-52.2), followed by disease (31.4%, 157/500, 95%CI 27.5-35.6) and fractures (12.4%, 62/500, 95%CI 9.8-15.6).

Between 1996 and 2013, 1297 stranded and rehabilitated turtles died unassisted while in care. The highest number of these turtles died for unknown reasons (63%, 818/1297, 95%CI 60.4-65.6), followed by disease (23.8%, 309/1297, 95%CI 21.6-26.2) and fractures (4.8%, 63/1297, 95%CI 3.8-6.1).

Between 1996 and 2013, 1173 stranded and rehabilitated turtles were released from rehabilitation alive. The number of turtles which were released for each stranding cause was variable, including unknown reasons (47.4%, 556/1173, 95%CI 44.5-50.3), buoyancy disorder (33.5%, 393/1173, 95%CI 30.8-36.2), and disease (5.4%, 64/1173, 95%CI 4.3-6.9).

4.4.2.2. *Time in Care for euthanized, died unassisted and released rehabilitated turtles*

2494 stranded and rehabilitated turtles had duration of care recorded (84% of all animals admitted to rehabilitation), 1139 of which died unassisted in care (45.7%), 480 were euthanized (19.2%) and 875 were released (35.1%). When analysing duration of care across all outcomes this was compared as a combined total of turtles sent to rehabilitation.

Table-A-1 shows the grouped duration in care before outcome, with 6.6% (165/2494, 95%CI 5.7-7.7) of all turtles were released within the first 7 days, 4.2% (106/2494, 95%CI 3.5-5.1) of all turtles were released between days 7 and 28, and 24.2% (604/2494, 95%CI

22.6-26.9) were released after 28 days. Since 1999, the average days in care before release decreased from 392 to 84 days, but the minimum days in care before release remained low at an average of 0.74 of a day (**Table-A-1**).

Table-A-1 shows the changes over the years in duration in care over the years; the average number of days in care before being euthanized varied over the years, with an overall decrease. 11.9% (298/2494, 95%CI 10.7-13.3) of all turtles were euthanized in the first 7 days, 3.9% (98/2494, 95%CI 3.2-54.8 of all turtles were euthanized between 7 and 28 days, and 3.3% (84/2494, 95%CI 2.7-4.1) of all turtles were euthanized after 28 days (**Table-A-1**).

Between 1996 and 2013, 25.4% (634/2494, 95%CI 23.7-27.2) of all turtles died without assistance within the first 7 days, 11.3% (283/2494, 95%CI 10.1-12.6) died between days 7 and 28, 8.9% (222/2494, 95%CI 7.8-10.1) died after the first 28 days (**Table-A-2**). In 1997 there was a spike in the average number of days in care before unassisted death after which the average days in care before death occurred decreased from 41-15 days.

4.4.3. Recaptures of turtles sent to rehabilitation

Between 1996 and 2013 of the 1173 turtles released from rehabilitation, 101 turtles were recaptured **(Table 4.4)**. This represented 8.6% (101/1173, 95% CI 7.1-10.4) of the turtles released from rehabilitation.

Table 4.4. Original reason for stranding, with subsequent recapture reason and fate.

Original cause of stranding	Subsequent recapture method	Alive	Alive and subsequently died unassisted	Dead	Euthanized	Grand total
Fractures		**3**	**0**	**0**	**0**	**3**
	Unknown	1	0	0	0	1
	Rodeo	1	0	0	0	1
	Nesting	1	0	0	0	1
Depredation		**1**	**1**	**0**	**0**	**2**
	Unknown	1	1	0	0	2
Disease		**5**	**0**	**0**	**2**	**7**
	Disease	1	0	0	1	2
	Buoyancy Disorder	1	0	0	0	1
	Unknown	2	0	0	1	3
	Rodeo	1	0	0	0	1
Entanglement Crabbing		**1**	**0**	**0**	**0**	**1**
	Nesting	1	0	0	0	1
Entanglement fishing		**2**	**0**	**1**	**0**	**3**
	Unknown	2	0	1	0	3
Entanglement rope		**1**	**0**	**0**	**0**	**1**
	Rodeo	1	0	0	0	1
Buoyancy Disorder		**22**	**4**	**4**	**3**	**33**
	Fracture	1	0	0	0	1
	Disease	3	1	0	1	5
	Entanglement fishing	2	0	0	0	1
	Buoyancy Disorder	3	0	0	0	3
	Unknown	9	3	4	2	18
	Rodeo	4	0	0	0	4

Table 4.4 Continued

SCP		**5**	**0**	**0**	**0**	**5**
	Entanglement fishing	1	0	0	0	1
	SCP	4	0	0	0	4
Unknown		**17**	**8**	**15**	**6**	**46**
	Disease	0	1	0	4	5
	Buoyancy Disorder	2	0	0	0	2
	Unknown	7	7	15	2	31
	Rodeo	8	0	0	0	8
Grand Total		**57**	**13**	**20**	**11**	**101**

Of the turtles released from rehabilitation and subsequently recaptured, 76.2% (77/101, 95%CI 67.1-83.4) were recorded as restranded and 17.8% (18/101, 95%CI 11.6-26.4) of them were recorded during normal population studies. The remaining six turtles 5.9% (6/101, 95% CI 2.7-12.4) recaptured during fishing activities.

Twenty of the turtles that subsequently restranded were dead (1-2820 days after release, average 485 days ± 725), 11 were alive and subsequently euthanized (1-2534 days after release, average 446 days ± 720), 13 were alive and subsequently died unassisted (1-1619 days after release, average 356 days ± 517) and 33 were alive and re-admitted for rehabilitation (1-1130 days after release, average 225 ± 284).

4.4.3.1. ***Recapture/Restranding Cause compared to original stranding cause***

The most common original cause of stranding for turtles that were recaptured alive was buoyancy disorder (39.6%, 22/57, 95%CI 27.1-51.2); followed by unknown original cause of stranding (29.8%, 17/57, 95%CI 19.5- 42.6); all other original causes contributed a total of 31%. For the nine identified categories of restranding; seven of these categories (depredation, disease, entanglement in fishing gear, entanglement in rope, buoyancy disorder, shark control program (SCP), and unknown causes) showed there was more than a 50% chance the turtle would restrand for the same reason as it originally stranded **(Table 4.4).**

4.4.4. Cost of rehabilitation

Collectively, SeaWorld, Australia Zoo Wildlife Hospital and Underwater World-SEA LIFE Aquarium reported spending a total average of $AUD112, 000 per annum on marine turtle rehabilitation to treat approximately two thirds (68%) of all marine turtles received for rehabilitation. Extrapolating from the annual amount spent at SeaWorld, Australia Zoo Wildlife Hospital and Underwater World-SEA LIFE Aquarium and assuming that other facilities have a similar expenditure, this approximates $AUD165, 000 being spent per annum in Queensland across the participating rehabilitation facilities on marine turtle rehabilitation.

Over the 18 years of this study, 2970 turtles were admitted to all rehabilitation facilities in Queensland, equating to an average of 165 turtles admitted per annum, costing approximately $AUD1,000 per animal admitted if you average total money spent on marine turtle rehabilitation per animal admitted. Over the 18 years of this study, 1173 of these turtles were released, equalling approximately 65 turtles per year at an estimated cost of $AUD 2,583 per animal released from rehabilitation if you average total money spent on marine turtle rehabilitation per animal released. Over the 18 years of this study, 101 of these released turtles were recaptured, equalling an average of 5.6 turtles per year at a costing on average $AUD 29,464 to rehabilitate each animal that is caught again. Of all of these animals admitted across all rehabilitation facilities for marine turtles, rehabilitated, released and recaptured, only 24 turtles were recaptured as functioning healthy members of the wild population, equalling approximately 1.3 turtles per year at a cost of $AUD 123, 750 to return a single animal to the functional population.

When analysing the costs for animals which were not successful during rehabilitation it is estimated that approximately $AUD 1650 is spent per turtle which is either euthanized or dies while in care.

4.5. Discussion

This investigation found that different causes of stranding influenced the survival for individuals, in terms of length of time in care, and survival of rehabilitation and post rehabilitation success. This provides rehabilitation centres with important information about resource outlay, particularly if success rates are poor (approximately 8.6%).

When analysing stranding data for Queensland between 1996 and 2013, Flint et al. (2015) found several significant trends in stranding numbers including: (i) an increase in the number of turtles reported stranded in Queensland during the study period (R^2= 0.6377); (ii) a species (loggerhead and green turtles (77.4%)) prevalence; (iii) a seasonal effect on different age classes, with most overall strandings occurring between August and November (47%); and (iv) stranding hotspots (Moreton Bay, Hervey Bay, Rockhampton region and Cleveland Bays) persisting throughout the study timeframe. These hotspots correspond to major freshwater discharge points as well as highly developed/populated areas.

Green turtles were the most frequent species and immature turtles were the most frequent age class sent to rehabilitation both by numbers and proportion, most likely because green turtles represent the largest proportion of the Queensland marine turtle populations and small immature turtles are the largest cohort of this population (Chaloupka, 2002a; Chaloupka and Limpus, 2001). Further, small immature turtles are likely to be the most susceptible cohort as a result of having a naïve immune system (Flint et al., 2010c, 2010d) to numerous potential stressors and being obligate residents of nearshore habitats that may be subject to a range of environmental stressors (Flint et al., 2010c, 2010d; Limpus et al., 1994b). The larger number of small immature being found is further compounded with them being nearshore residents, which has been shown to increase the likelihood of recorded strandings (Peltier et al., 2012).

Green turtle stranding increased at a rate of 9.9% per annum (pa) over the study period. However, during a similar timeframe, the southern Great Barrier Reef green turtle population increased at a rate of approximately 10.6% pa (Chaloupka and Limpus, 2001) As well, the southern Great Barrier Reef foraging loggerhead turtle populations declined over this same period at approximately 3% pa (Chaloupka and Limpus, 2001) and stranding rates decreased at a rate of 3.5% pa. Stranding numbers in Queensland may be a normal function of the population and a proxy for the overall population change.

For stranded turtles which were recaptured, the primary known presenting problem or disorder were buoyancy disorders (35%), disease (8%) and fractures (trauma)(1%) **(**Figure 4.3**).** As the former two signs may both represent multiple conditions, successful treatment within rehabilitation centres requires the determination of the cause of stranding diagnosis/underlying health problem. Potentially as a result of this limitation, the most common causes of restrandings were turtles that originally presented with one of these two conditions. Conversely, rehabilitation centres traditionally have a good success rate identifying and treating trauma for complete release with very few of these turtles restranding. Further, Flint et al., (2015) suggested there are certain times of the year when it can be expected certain conditions are going to present. Disease and buoyancy disorders were highest at the end of winter, likely when resources were stressed and immune systems were under duress. Trauma was most prevalent during the summer months when recreational boating may be at its greatest as seen in other popular urbanized embayments (Widmer and Underwood, 2004). Despite the introduction of "go-slow zones" for motorized crafts there has been an overall increase in the proportion of turtles admitted to rehabilitation with fractures. This may not just result from the obvious threat of recreational or commercial boating but could include: (i) more turtles sustaining non-life threatening injuries enabling them to survive and be taken to rehabilitation; (ii) an increase in public awareness increasing the number of animals being taken to rehabilitation; or (iii) the population of turtles is increasing (Chaloupka and Limpus, 2001), increasing the likelihood of a negative encounter with a recreational vessel. In all cases, trauma may be reduced by improved restrictions in certain zones on a seasonal basis; such as has been successfully employed to protect the Florida Manatee (*Trichechus manatus latirostris*) in high use areas (Calleson, 2014; Calleson and Frohlich, 2007).

For a short period between 2005 and 2009, euthanasia rates increased across rehabilitation centres in Queensland **(Figure 4.3)**. As there were no recorded epidemics during this time, reasons for this trend remain unclear. There was no significant shift in expertise during this time and the majority of new rehabilitation centres opened after the 2010 major floods (Meager and Limpus, 2012b). A potential for this peak in 2009 is an oil spill which occurred in the northern Moreton Bay Area (SEQ Catchments, 2011) but this does not account for the 4-year period prior to this catastrophe. Funding to individual rehabilitation centres or recommended treatment regimens may have influenced this peak.

During rehabilitation, over 25% of turtles die unassisted during the first week of treatment (**Table-A-2**), suggesting progression of cause of stranding is too advanced or the disease syndrome is too complex for treatment and successful reintroduction to the population. This phenomenon must be addressed to ensure diagnostic regimes, animal welfare and limited resources are being optimally used. Anthropogenic causes (not including fractures) were least successfully treated. A likely reason is the degree of intervention required when compared with diseases that can be systemically treated with appropriate palliative care **(Table 4.3)**.

Some of the results found in this study differ from that of Bosagna, (2012), who found that only 7.14% of turtles admitted to rehabilitation died unassisted in the first 15 days of treatment. Possible reasons for this are that Bosagna, (2012) only analysed 2 years of data from three rehabilitation facilities. The results for turtles that were released and turtles which were euthanized were similar when the initial time periods are compared (<15 days (Bosagna, 2012), and >7 days (this study).

Similar rehabilitation results were obtained by Baker, Edwards & Pike, (2015) who looked at the outcome of turtles from rehabilitation in Florida. They found that 36.8% of turtles admitted to rehabilitation were released back into the wild, while 55.3% died while in rehabilitation. This suggests treatment regimens and approaches and stressors between the USA and Australia may be comparable.

It is difficult to assess the true success of rehabilitation without following each individual. There have been few studies investigating the ability of turtles to readapt to the wild after rehabilitation (Cardona et al., 2012; Feck and Hamann, 2013; Tomás et al., 2001). The two most appropriate methods for assessing post-rehabilitation survivorship are satellite-tracking individuals or tagging individuals and monitoring for their restranding or recapture with time. Queensland has provided an ideal opportunity for a case study of this issue because of the long running programs of both stranding and routine population monitoring. Shimada et al., (2016) analysed satellite tracking data of turtles which had been displaced from their original capture site (inferred home area). Of the 59 displaced turtles, 52 returned to their home areas. All 52 non-displaced turtles remained within their home areas. This indicates that turtles which are removed from their home area are likely to return to that location. It follows that if turtles were exposed to threats at the original location, once they return from rehabilitation they will possibly be exposed to those same threats again and hence potentially succumb and restrand. This study showed the original cause of stranding is closely linked to the cause of restranding suggesting either incomplete treatment during rehabilitation (e.g. not eliminating the disease during treatment) or re-exposure or behavioural predisposition in certain turtles to recreate the hazard (e.g. SCP or fishing line entanglement).

The number of turtles reported stranded during this study represent approximately <0.001% ((9641 turtles/18 years)/641262) of the suspected benthic southern Great Barrier Reef population (Chaloupka, 2002b). Despite the successes and challenges of rehabilitation, a large number of turtles were released over 18 years, from which 9% were recaptured either as restranded or healthy and part of normal survey activities giving an insight in to the long-term outcome of human intervention as a population conservation tool. To put this in context, functional population recapture rates range from 8%-84.3% depending on age class (Bell et al., 2012; Chaloupka and Limpus, 2005, 2002). While this may indicate a large proportion of released turtles are subsequently dying or otherwise being removed from the population, this may not necessarily be the case because some turtles may be released back into an area that is not within regularly sampled regions of those serially surveyed.

Based on state-wide recapture data and the cost of rehabilitation at the three main rehabilitation centres in Queensland (Australia Zoo Wildlife Hospital, SeaWorld and Underwater World-SEA LIFE Aquarium), only 1.3 turtles are successfully returned to the functional population each year at a cost of $AUS123,750 an animal. Even though these costs should only be viewed as indicative estimates as opposed to a true calculation per known animal released, the pattern indicates this may not be economical and these high costs and low numbers are likely not contributing to conserving a population (Baker et al., 2015; Cardona et al., 2012; Feck and Hamann, 2013; Moore et al., 2007; Quakenbush et al., 2009). However, rehabilitation may still benefit the populations through public education, increased awareness and advances in our understanding and treatment of the diseases and biology of marine turtles through this means of conservation. With the high costs associated with unsuccessful cases admitted through rehab, there may be more productive way some of this money could be used to educate the public so that more marine turtles in the wild can benefit (such as boater education, beach protection/monitoring, anti-litter campaigns, fisher education). Similarly, scientist education and development of better protocols and understanding of the underlying processes causing stranding may benefit from funding.

There is no question rehabilitation plays an important role in the care of individual turtles by reducing suffering and treatment of certain conditions. Part of the rehabilitation process can include euthanasia as a treatment option to prevent individual suffering and can add value to research through appropriate post-mortem investigations. Rehabilitation also provides a valuable vehicle for public education and conservation messages which, in turn, increase public awareness and hopefully reduce anthropogenic causes of stranding (Baker et al., 2015; Cardona et al., 2012; Feck and Hamann, 2013). This rationale adheres to the One Health paradigm, whereby educating people to take more responsibility for their actions can reduce their impacts (by changing perception of how they treat the ocean) that can have a direct impact on the environmental health and resultant animal health (Flint, 2013; Schwabe, 1969). However, with respect to augmenting the population of marine turtles, rehabilitation may not contribute to survivorship. This is influenced by the size and robustness of the local turtle population, the factors affecting stranding and the conservation status of the local population.

Given the prevalence of: (i) certain cohorts; (ii) seasonality; (iii) certain causes of stranding; and (iv) higher numbers of strandings at four locations along the Queensland coastline (hotspots), it may be more appropriate to direct rehabilitation efforts to events of higher demand. For example, this could mean creating MASH (Mobile Army Surgical Hospital)-like response centres that target their care to immature turtles that present at the end of the winter period (August-September) - to treat/evaluate boat strike and unknown causes, within the recognized stranding ‘hotspots’. Focused triage and treatment may represent a significant cost saving to rehabilitation centres throughout Australia. As 22% of animals which are admitted to rehabilitation reach an outcome within 7 days, the creation of such response centres, will allow turtles with obvious disorders which only need short term immediate care to be treated and released, freeing up resources and space in rehabilitation centres for turtles which require more in-depth/long-term care.

Despite the costs involved, rehabilitation continues to be a tool for conservation because it provides a platform to educate members of the public about threats to marine turtle survival (Addison and Nelson, 2000; Feck and Hamann, 2013). It has been shown that when people visit zoos or aquaria that have a prominent conservation message, the visitors’ mindsets can be changed towards being more pro-conservation (Adelman et al., 2000; Falk et al., 2007; Wyles et al., 2013). Rehabilitation also provides insight into the diseases causing stranding through ancillary investigations (Flint et al., 2010d). Information from post-mortem investigations such as necropsy can help first-responders to gather insight into what disease and parasitic prevalence may be present during normal times to create a baseline of “background” pathologies for a region. This in turn may aid in determining when a syndrome becomes an outbreak (or unusual mortality event) and allows future first-responders to be better prepared.

This article is a desktop analysis and does not contain any studies with animals.

We would like to thank the Queensland Marine Wildlife Strandings and Mortality network and all contributors to the StrandNet database. We would also like to thank turtle rehabilitation facilities staff at Underwater World-SEA LIFE (Mooloolaba, QLD), Australia Zoo (Beerwah, QLD and SeaWorld (Gold Coast, QLD) for providing access to their data. We would also like to thank Dr. Jeffery Miller for his review of this document.

Chapter 5. What happens to stranded turtles not sent to rehabilitation?

This chapter analyses the data recorded within StrandNet of animals that were stranded alive but were released *in situ*. This chapter provides a complementary analysis to the previous chapter but compares it to animals that received no or minimum care. With the exception of the previous chapter, this provides new study combining StrandNet data and the Queensland Turtle Conservation Project data.

This article has been submitted to Marine Turtle Newsletter.

5.1. Introduction

As awareness in wildlife conservation and welfare has increased around the world, there has been a corresponding increase in the numbers of stranded marine animals that are reported to government authorities by a host of stakeholders, including special interest groups and members of the public, with the intent the stranded animal will be rescued and returned to the ocean (Feck and Hamann, 2013). In places like Queensland, a good proportion of these animals that strand alive are sent to rehabilitation centres for medical assessment and treatment. However, there are a number of these animals that are triaged and deemed fit for release directly back into the wild by experienced first responders (rangers and biologists) using protocols that have been refined over and taught for the last few decades. For example, of the 13854 marine turtle strandings recorded along the Queensland coastline between 1996 and 2013, 5491 (39.6% of all stranded) of these animals stranded alive, 2970 (54% of alive) of these animals were sent to rehabilitation facilities and 2052 (37% of alive) were left *in situ* or directed back into the water.

We examined the fate of turtles that stranded alive along the Queensland coastline between 1996 and 2013 that were returned to the ocean after initial field triage and were not sent to rehabilitation. For this study we also included animals that had been encountered during the Shark Control Project (SCP) and released directly without triage. The study area used for this analysis was between latitude -10.78° to -28.16° and longitude 142.15° to 155.00° encompassing the timeframe from the 1st of January 1996 to the 31st of December 2013.

The release rate of rehabilitated marine turtles for the Queensland coastline between 1996 and 2013 has been calculated to be 39% of all animals admitted to rehabilitation (Flint et al., 2017a). The amount of treatment received in Australian triage locations varies depending on the available resources. Turtles not admitted into rehabilitation facilities are returned directly to the water for a variety of reasons including the minor nature of the trauma/injury or the accessibility and resources available to receive and treat the animals (Flint et al., 2017a). What has not been determined until now is the success rate of only providing initial field triage and returning the turtle to the water without the intensive assessment, treatment and associated financial costs of rehabilitation.

The term *stranding* is used to incorporate all reported sick, injured, incapacitated or dead marine turtles that either were found washed ashore or encountered at sea. It includes turtles which were entangled in synthetic debris including fishing nets and line, as well as turtles which were rescued (Biddle and Limpus, 2011; Flint et al., 2015; Geraci and Lounsbury, 2005; Meager and Limpus, 2012a). For each animal, a single primary cause of stranding was recorded.

5.2. Methods

Stranding and release information was obtained through StrandNet, the Queensland Government's Department of Environment and Heritage Protection state-wide database reporting threatened stranded marine animals and the Queensland Turtle Conservation Project (QTCP) database (Flint et al., 2017a, 2015). Animals from the QTCP database were used if their coding within the database indicated that they had been rescued or subsequently died after being released. For this study it also included animals which were encountered during the SCP that were not returned to land for assessment. Animals were matched between the databases using unique identifiers. To find subsequent recaptures of the same animal queries were done with the capture date that was greater than the first recorded stranding.

The date a turtle stranded was used as a proxy of time of incident, and was grouped by month for general analyses, as described by Flint et al. (2015).

For each animal, a single primary cause of stranding was presented and is based on summation of gross examination, photographs and/or necropsy performed by trained personnel (Flint et al., 2015). It is acknowledged that animals often present with multiple disorders, but StrandNet only records the suspected primary cause of stranding or most obvious condition (Flint et al., 2017a); which may create a limitation by not acknowledging concurrent, insidious or obscure pathologies. The cause of stranding identified in StrandNet was then used to group the reasons for stranding into 18 categories. These categories are: buoyancy disorders, courting related rescues, depredation, disease, dredging, entanglement ghost fishing, entanglement crabbing, entanglement fishing, entanglement rope, fractures, ingestion of foreign material, land reclamation, nesting beach related rescues, netting, other anthropogenic, SCP, unknown and unknown natural.

Records as to the specific treatment provided are sparse. However, from the information available, in most cases animals received little or no treatment. They were removed from the threatening process (e.g. released from fishing line, rescued from disorientated situations) and immediately directed back towards the water.

Anthropogenic (other) are any cause of stranding which is anthropogenic in nature that is not previously described including fishing entanglement and ingestion of foreign objects.

Buoyancy disorder is used to describe turtles which were observed floating and in which no other presenting problem or disorders were observed.

Fracture is used to denote any form of fracture to a turtle that is attributable to anthropogenic causes (e.g. boat strike or blunt force trauma).

Disease is classified as turtles which exhibit protracted ill health from a cause consistent with a physiological condition and not otherwise caused by anthropogenic activity (e.g. fracture, entanglement). This is often linked with poor body condition.

Nesting beach rescues are nesting females which were rescued while undertaking normal nesting activities. They may have been rescued from anthropogenic (e.g. altered light horizons, holes in sand) or natural causes (e.g. trapped in vegetation or among rocks, fallen on upside down).

Courtship Rescues are courting animals which were rescued while undertaking normal courtship activities. They may have been rescued from being flipped in the surf or other natural causes.

For this study, *survivorship* is defined as a turtle being found in good condition at least once after release from rehabilitation. Determination of condition were made based on coming onto a nesting beach or laparoscopic examination of the gonads or via in-water population surveys and being found to be in good condition at each capture (Flint et al., 2010c; Limpus and Reed, 1985a). Release was deemed non-successful if the animal was reported stranded again (either dead or alive) within the timeframe of the collected dataset (1996-2013).

RWR – “Recapture without rehabilitation” is where turtles were released without rehabilitation and were then recaptured at a later date.

Rodeo is a technique developed to capture turtles during in-water surveys. This technique is presented pictorially in Limpus (1978) and described in depth in Limpus (1985).

R was used to perform all statistical analysis (R Core Team, 2016). Results were presented as descriptive statistics with rigor expressed as a standard 95% confidence interval. Confidence intervals were selected as they represent the variance within a dynamic population.

5.3. Results and Discussion

Along the Queensland coastline between 1996 and 2013 there were 14334 marine turtles recorded as stranded or involved in the SCP, of which 5491 were reported as stranded alive. Of the animals which stranded alive 45.9% (2520/5491, 95% CI 44.6-47.2%) were not sent to rehabilitation.

Table 5.1 shows the outcomes of marine turtles which are not admitted to rehabilitation in Queensland.

Table 5.1 Outcomes of turtles not admitted to rehabilitation in Queensland, n=2903.

Outcome	**Number**	**Proportion of animals which strand alive and were released**
Animals released no rehab	1729	59.56%
Animals which stranded alive but died without admitted to rehab	602	20.74%
Animals euthanized by first responders	189	6.51%

Table 5.2 shows the causes of stranding for turtles which strand and were not sent to rehabilitation.

Table 5.2 Original cause of stranding for animals which stranded alive and were released and also animals which stranded alive and subsequently died.

Cause of stranding	Number stranded alive and released	Number stranded alive and subsequently died
Buoyancy disorder	303	0
Courting related rescues	9	0
Depredation	8	12
Disease	58	173
Dredging	0	8
Entanglement ghost fishing	27	0
Entanglement crabbing	84	8
Entanglement fishing	35	3
Entanglement rope	5	1
Fractures	7	89
Ingestion of foreign material	1	29
Land reclamation	34	0
Nesting beach related rescues	117	0
Netting	8	1
Other anthropogenic	25	9
SCP	492	2
Unknown	510	451
Unknown natural	6	5
Grand Total	**1729**	**791**

For the turtles that stranded alive and were subsequently released without rehabilitation, 429 were recaptured (25.8%, 429/1729, 95%CI 9.4-12.9%).

Table 5.3 compares the original cause of stranding with the subsequent recapture method and status.

Table 5.3. Original cause of stranding compared to recapture method and status. (RWRs). Bold numbers under the Recapture status are totals for all recaptures under each particular original cause of stranding.

Original cause of stranding	Recapture Method	Recapture Status				
		Alive	Alive subsequently died	Dead	Euthanized	Total
Buoyancy Disorder		**1**	**1**	**0**	**0**	**2**
	Disease	0	1	0	0	1
	Unknown	1	0	0	0	1
Courting		**9**	**0**	**0**	**0**	**9**
	Courting	4	0	0	0	4
	Nesting	5	0	0	0	5
Depredation		**0**	**0**	**1**	**0**	**1**
	Unknown	0	0	1	0	1
Disease		**1**	**0**	**1**	**0**	**2**
	Rodeo	1	0	0	0	1
	Unknown	0	0	1	0	1
Entanglement Crabbing		**2**	**0**	**1**	**0**	**3**
	Buoyancy Disorder	1	0	0	0	1
	Rodeo	1	0	0	0	1
	Unknown	0	0	1	0	1
Entanglement Fishing		**4**	**0**	**1**	**0**	**5**
	Nesting	1	0	0	0	1
	Rodeo	2	0	0	0	2
	SCP	1	0	0	0	1
	Unknown	0	0	1	0	1
Fractures		**1**	**0**	**1**	**0**	**2**
	Nesting	1	0	0	0	1
	Unknown	0	0	1	0	1

Table 5.3 continued

Land Reclamation	**4**	**0**	**0**	**0**	**4**
Land Reclamation	4	0	0	0	4
Nesting Beach	**66**	**0**	**1**	**0**	**67**
Nesting	66	0	0	0	66
Unknown	0	0	1	0	1
Netting	**0**	**0**	**1**	**0**	**1**
Unknown	0	0	1	0	1
Other Anthropogenic	**9**	**0**	**0**	**0**	**9**
Nesting	9	0	0	0	9
SCP	**293**	**1**	**7**	**0**	**301**
Nesting	5	0	0	0	5
SCP	285	1	7	0	293
Observation	2	0	0	0	2
Fishing	1	0	0	0	1
Unknown	**17**	**1**	**1**	**3**	**22**
Disease	1	0	0	0	1
Nesting	4	0	0	0	4
Rescue	0	0	0	1	1
Rodeo	6	0	0	1	7
Unknown	6	1	1	1	9
Unknown Natural	**1**	**0**	**0**	**0**	**1**
Rodeo	1	0	0	0	1
Grand Total	**408**	**3**	**15**	**3**	**429**

Figure 5.1 shows the demographics of RWRs.

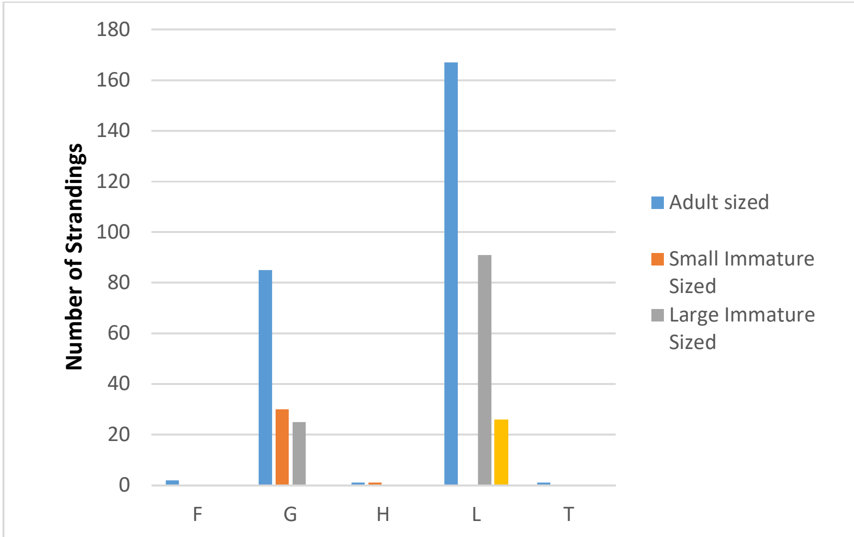


Figure 5.1 Demographics of RWRs. F = Flatback turtles, G= Green Turtles, H = Hawksbill turtles, L= Loggerhead turtles, T= unidentified species.

RWR's were more likely to be encountered during the summer months (December-January), with the lowest number occurring during March.

When analysing the number of records of RWRs originally caught in the SCP (n=301), there were 67 animals that were caught more than once. It is of particular note that 15 animals were caught five or more times (n=221 records).

We evaluated the effectiveness of triage and release as a conservation tool. We found that different causes of initial stranding influenced the subsequent recapture method/cause.

Loggerhead turtles were the most frequent species to strand and be released without being admitted to a rehabilitation facility (**Figure 5.1**).

For animals which were stranded and released without rehabilitation, the majority of animals were stranded for unknown reasons (**Table 5.2**). For known causes, the majority were SCP and buoyancy disorder. This indicates that more work needs to be undertaken into identifying the apparent unknown reasons; for example, determining whether first responders need more training, or more investigations are needed to determine the less obvious causes of stranding.

For RWRs, the primary initial problem or disorder was SCP, followed by nesting beach rescues and unknown causes (**Table 5.3**). From our recorded data, animals which were initially caught within the SCP were encountered again within the SCP (**Table 5.3**). This is a potential problem for turtles since, depending on capture method, they appear to be "reoffenders" to this hazard **(Table 5.4**). This phenomenon needs further investigation as to the reasons behind them returning, this could include investigations to determine if turtles are seeking food or if it is accidental hooking as turtles are swimming around. Capture method influenced the survival and recapture of animals caught within the Shark Control Program, for example animals caught in nets have a lower chance of survival and hence a lower chance of recapture vs. animals which are hooked on hooks (C.J Limpus pers comms).

Table 5.4. Number of RWRs originally caught in the SCP and the number of recaptures within the SCP.

Number of Recaptures	**Number of animals per number of recaptures**
2	31
3	6
4	15
5	3
6	3
7	1
8	1
9	1
12	1
18	1
24	2
40	1
46	1
Grand Total	**67**

Animals which stranded for unknown reasons were half as likely to be encountered as part of the healthy population, as they were to strand again (**Table 5.3**). If resources are available, first responders need some further guidance when dealing with these animals as a proportion may need the more intensive diagnoses to accurately treat and care for them that is available at rehabilitation facilities. If the cause of stranding is not obvious, it may be prudent to assume the animal falls under this higher care required category. There is also a need for further investigation into the unidentified causes of stranding to understand exactly what the range of causes are and the potential treatments/mitigations available in the field.

When comparing the recapture of animals released from rehabilitation, 2% of animals released from rehabilitation are recaptured as part of the healthy population (Flint et al., 2017a). This compares to the 25.2% of animals which were recaptured as part of the healthy population after being released back into the ocean without rehabilitation (**Table 5.3**). While on the surface this suggests recognizing minor issues and returning marine turtles to the ocean post triage provides a 10-fold increase in the likelihood of returning to the healthy functional population, rehabilitation plays an important role in the care of individual animals and seriously ill animals. It also acts as a vehicle for public conservation education with the ultimate goal of minimizing anthropogenic impacts on marine turtles (Baker et al., 2015; Cardona et al., 2012; Feck and Hamann, 2013; Flint et al., 2017a).

A limitation to this study was the inclusion of SCP captures and nesting females, which are healthy animals potentially re-exposed to the same threat numerous times. These two categories of stranding represented 368 of the 429 RWRs, biasing findings. Removing both from calculations showed all other categories of stranding and release without rehabilitation to have a 59% chance (30-fold increase to rehabilitated animals) of returning to being part of the functional healthy population. However, larger datasets are required to uphold or reject this finding.

This study also does not take into account the fact that some animals strand in remote locations where additional limitations exist, such as inability to transport to a rehabilitation facility. Despite this limitation, it does not take away from the fact that animals which are left to "natural process" do not appear to have a significantly reduced chance of survival or restranding compared with animals which are sent through rehabilitation.

First responders have an important role in the triage of marine turtles which strand and when appropriate triage is applied, the time, animal stress and expense of rehabilitating turtles can be avoided. There is still some room for knowledge improvement in terms of knowing when animals need more intense treatment and when they can be released without being admitted to a rehabilitation facility.

5.4. Acknowledgements

We would like to thank the Queensland Marine Wildlife Strandings and Mortality network, EHP and all contributors to the StrandNet database.

This study was partially funded by the Australian Research Council Linkage Project Grant LP110100569.

Chapter 6. The impact of environmental factors on marine turtle stranding rates

This chapter provides analysis between marine turtle strandings and environmental factors. Prior to this study there was anecdotal evidence that marine turtle strandings were affected by environmental variables. This analysis looked at the time lags between the environmental variables and the response by marine turtle strandings. This study provided a baseline study for the use in the creation of predictive models.

This paper has been published in PLoS ONE.

6.1. Abstract

Globally, tropical and subtropical regions have Queensland has experienced an increased frequency and intensity in extreme weather events, ranging from severe drought to protracted rain depressions and cyclones, these coincided with an increased number of marine turtles subsequently reported stranded. This study investigated the relationship between environmental variables and marine turtle stranding. The environmental variables examined in this study, in descending order of importance, were freshwater discharge, monthly mean maximum and minimum air temperatures, monthly average daily diurnal air temperature difference and rainfall for the latitudinal hotspots (-27°, -25°, -23°, -19°) along the Queensland coast as well as for major embayments within these blocks. This study found that marine turtle strandings can be linked to these environmental variables at different lag times (3-12 months), and that cumulative (months added together for maximum lag) and non-cumulative (single month only) effects cause different responses. Different latitudes also showed different responses of marine turtle strandings, both in response direction and timing.

Cumulative effects of freshwater discharge in all latitudes resulted in increased strandings 10-12 months later. For latitudes -27°, -25° and -23° non-cumulative effects for discharge resulted in increased strandings 7-12 months later. Latitude -19° had different results for the non-cumulative bay with strandings reported earlier (3-6 months). Monthly mean maximum and minimum air temperatures, monthly average daily diurnal air temperature difference and rainfall had varying results for each examined latitude. This study will allow first responders and resource managers to be better equipped to deal with increased marine turtle stranding rates following extreme weather events.

6.2. Introduction

In recent years, tropical and subtropical regions, such as Queensland, have experienced many extreme weather events, including droughts, cyclones and protracted rain depressions. In Australia, during summer there is a heightened risk of extreme weather and warmer temperatures, the summer of 2010/2011 in Queensland is of particular note. During this time, cyclones and protracted rain depressions caused wide-spread flooding which in turn led to increased periods of turbid water and increased nutrient and sediment loads from freshwater run-off being dumped into all four major coastal waterways (Brisbane, Fitzroy, Burnett and Burdekin Rivers)(Devlin et al., 2012a). The cyclones and floods stressed coral reefs and seagrass beds causing large-scale die-off of ecologically important seagrass species and decreased water quality intermittently along the entire length of the Queensland coastline south from Cairns (Coles et al., 2012; Devlin et al., 2012a; Great Barrier Reef Marine Park Authority, 2011a; McKenzie et al., 2014). It was postulated that within a year (short-term) of these types of catastrophes, marine megafauna show an increase in the number of stranding, mortalities and exacerbated poor health conditions (Meager and Limpus, 2014). In a similar ilk, it has been shown that environmental variables affect seabird wrecks numbers and locations (Tavares et al., 2016).

The ongoing poor weather conditions recently experienced are unprecedented in the 35 year history of the Great Barrier Reef Marine Park (Great Barrier Reef Marine Park Authority, 2011a) and Queensland in general (Steffen et al., 2013) since European settlement. The magnitude and scale of the bad weather conditions experienced during early 2011 on the Great Barrier Reef have not been seen since recording began in 1918 (Great Barrier Reef Marine Park Authority, 2011a).

Norman et al.(2012) stated that the ability to understand and investigate marine mammal unusual mortality events and other unexpected strandings that involve substantial die-offs of the marine mammal population are important events which serve as indicators of ocean health. This may give better insight into larger environmental issues, which can have implications for human health and animal welfare. This One Health paradigm can also be applied to marine turtle strandings as marine turtles have been proposed as sentinels of environmental health (Aguirre and Lutz, 2004; Flint, 2013; Hamann et al., 2010) and, as such, an increase in the numbers of animals which strand can indicate that the environments in which they live have changed (Flint, 2013).

It has been suggested that marine turtle stranding numbers follow seasonal trends influenced by weather events as well as land-based and at-sea seasonal activities. There have been links made between extreme weather and increased strandings within 12 month periods as outlined by Marsh and Kwan, (2008); Meager and Limpus, (2014), (2012b) and Preen and Marsh, (1995).

Meager and Limpus (2012) stated that the most plausible explanation for the high rate of strandings and mortalities of nearshore green turtles during 2011 were extreme weather events that occurred in late 2010 and early 2011, which impacted seagrass foraging areas. They linked this because most of the examined mortalities were attributed to protracted ill health/poor body condition in green turtles and dugongs; which both primarily forage on seagrass. There was evidence that seagrass pastures, mangrove forests, algal beds and coral reefs in Queensland were impacted by a combination of elevated rainfall, flooding and three cyclones (Category 5 Yasi, Category 2 Anthony and Category 1 Tasha) with a protracted low pressure system during the summer of 2010/2011.

This study examined looked at marine turtle stranding rates in relation to certain environmental variables (including rainfall, freshwater discharge rates and air temperature). Different latitudinal blocks, species and age classes were investigated to determine if there were different responses. We summarized and analysed the available data to provide first responders and management agencies with information to better assist them when responding to stranding events. The databases used for this study are the most comprehensive databases available for Queensland marine turtle records and was established over 30 years ago.

6.3. Methods

6.3.1. Data

6.3.1.1. *Stranding Data*

StrandNet is the Queensland Government's Department of Environment and Heritage Protection (EHP) state-wide database which records dead, sick and injured threatened marine animals for the entire coast of Queensland and adjacent Commonwealth waters. Records are received from members of the public, and employees of EHP, Queensland Parks and Wildlife (QPWS), Queensland Department of Agriculture, and Fisheries (DAF) and the Great Barrier Reef Marine Park Authority (GBRMPA). Information is collated and stored in this central database. Once reports are entered by on-ground staff the information available is verified by regional and state coordinators for standardization.

Additional data is often obtained after the stranding event from veterinarians, pathologists and other biologists who complete more detailed post-mortem investigations.

6.3.1.1.1. *Biometrics (Age class, sex, species)*

As a proxy of age class, standard measurements such as curved carapace length (CCL) and tail to carapace length (TCL) were collected at the time of initial stranding (Limpus et al., 1994a). The breakdown of age class for loggerheads were adapted from Limpus et al., (1994b), hawksbills from Limpus, (1992) and other species were adapted from Limpus et al., (1994a).This data was used to assign turtles into 3 age classes: small immatures, large immature and adult sized.

Sex was determined by gonad examination by trained personnel either onsite or using photographs or measurements (Limpus and Limpus, 2003; Limpus and Reed, 1985a).

Based on dichotomous key characteristics (Environmental Protection Agency, 2008; Great Barrier Reef Marine Park Authority, 2007), species was determined as one of six turtle species including subspecies: green (*Chelonia mydas)*, loggerhead (*Caretta caretta)*, flatback (*Natator depressus*), hawksbill (*Eretmochelys imbricata)*, leatherback *(Dermochelys coriacea)*, olive ridley *(Lepidochelys olivacea)*, black turtle *(Chelonia mydas agassizi*), as a hybrid animal or species unknown. Due to debate over species versus subspecies and a small dataset, we removed the black turtle from the individual species analyses.

6.3.1.1.2. *Location*

The study area encompassed latitude -10.78° to -28.16° and longitude 142.15° to 155°. This part of the east coast of Queensland was selected as it has a long-term and comprehensive dataset; with data collection biased to regions of survey and higher populations. This limitation is openly acknowledged by Meager and Limpus (2012) but considered valid as a representative of a minimum recovery rate and indicative of trends occurring. As the exact location where a stranding was reported was not necessarily where the impact/incident occurred, strandings were grouped into latitudinal blocks of 1° to account for this potential error. The main areas of focus for this study were the hotspots recognized by Flint et al. (2015) as -27°, -25°, -23° and -19° (Figure 6.1). In addition, major embayments, irrespective of latitudinal blocks were assessed (Figure 6.2).

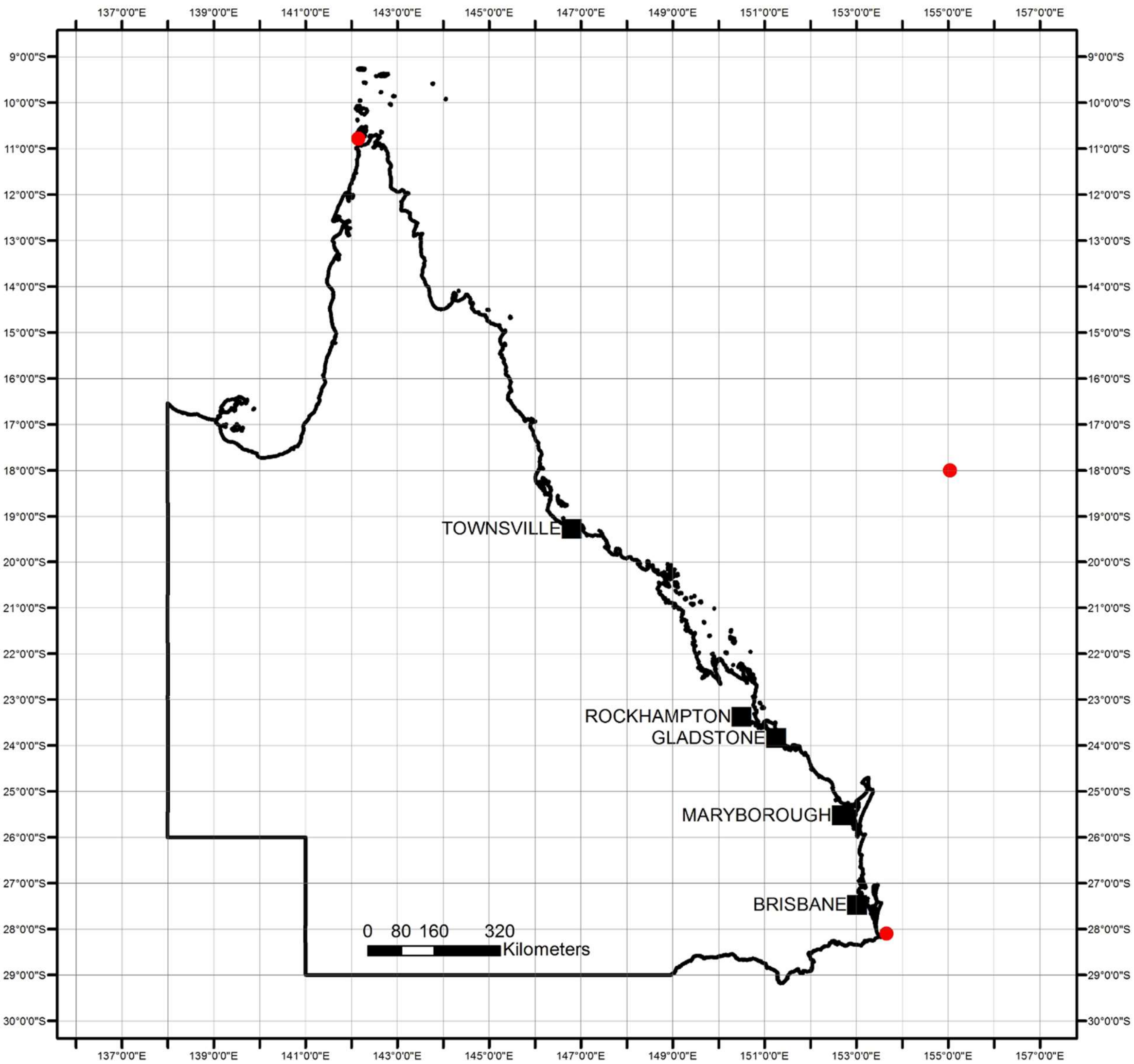


Figure 6.1 Map of Queensland coast. Red dots denote limits of study area.

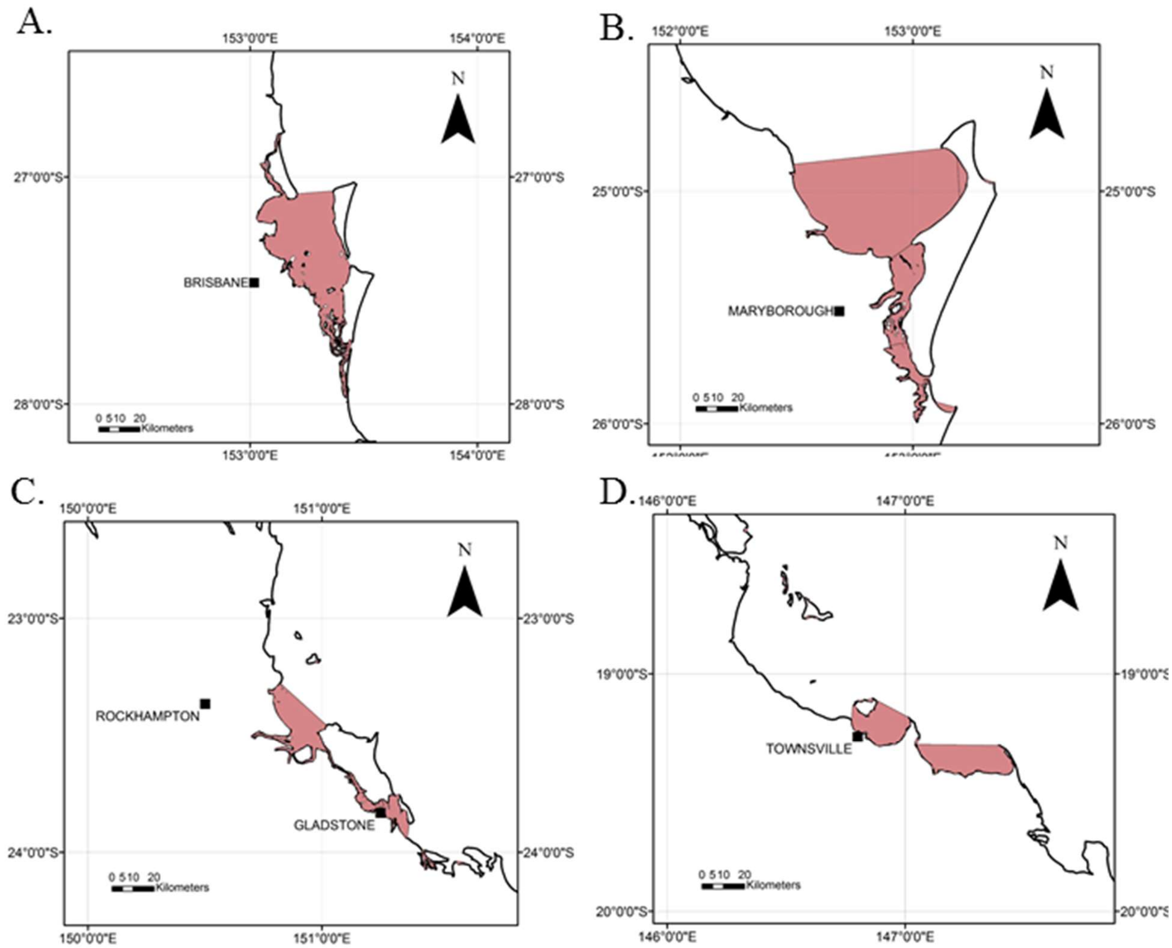


Figure 6.2 Embayments used for data analysis. A. 27°, B. 25°, C. 23°, D. 19°.

6.3.1.1.3. *Time*

The date a turtle was reported stranded was used as a proxy of time of death, grouped to a monthly scale.

6.3.1.1.4. *Cause of Stranding*

The term 'stranding' is used here to incorporate all reported sick, injured, incapacitated or dead marine turtles that were either found ashore or, in rare cases, were encountered at sea. It included turtles which were entangled in fishing nets, synthetic debris or rescued from a situation where they would have died had it not been for human intervention (Geraci and Lounsbury, 2005).

Within StrandNet, the primary cause of death/stranding was identified based on gross examination, photograph and/or necropsy by trained personnel (Meager and Limpus, 2014; Meager and Sumpton, 2016). The single cause of stranding identified in StrandNet was based on the summation of information available.

6.3.2. Environmental Data

Rainfall, freshwater discharge and air temperature were examined as environmental variables. These were selected as they provided the most comprehensive, readily available and up to date dataset of environmental conditions available. Turbidity, water temperature, pH and salinity were not used due to paucity of current available data along the Queensland coastline.

Freshwater discharge is the amount of freshwater running through a river's gauging (recording) station, measured in cumecs (cubic meter per second, $m^3.s^{-1}$). Freshwater discharge data was downloaded from the Department of Natural Resources and Mines (https://water-monitoring.information.qld.gov.au/) under the Creative Commons Attribution 3.0 Australia (CC BY) license. Discharge data from the most downstream gauging station for each major drainage area was grouped into 1° latitudinal blocks (27 stations for the 4 latitudes chosen). The discharge variables were then calculated for each latitude as follows: (1) peak discharge or maximum discharge in a given month across all stations; (2) monthly mean discharge across all stations; (3) cumulative mean for all stations across all stations. Data for each month between 1996 and 2013 was analysed (Meager and Limpus, 2014).

Rainfall and air temperature data was obtained from the Bureau of Meteorology for a central coastal station within each latitudinal block with a complete dataset. Mean monthly maximum and minimum air temperatures were used directly. The monthly average daily diurnal air temperature difference was calculated by obtaining the maximum and minimum daily air temperatures and calculating the difference, then averaging this value over the month. Data for each month between 1996 and 2013 was analysed.

6.3.3. Data Analysis

Data from StrandNet was grouped into 1° latitudinal blocks from the -28° (Queensland – New South Wales border) north to -16° (Cape Tribulation) for each month between January 1996 and December 2013 (Meager and Limpus, 2014). Only natural and unknown causes of death were used for this analysis, as anthropogenic causes can be seasonal due to increased activity (eg. Fishing and boating) (Meager and Limpus, 2014). The "unknown cause" used as the operating practice for StrandNet was applied when there was no obvious cause of trauma or subsequent analysis done (Meager and Limpus, 2014).

Strandings were also isolated from bays recognized from the Queensland Spatial Catalogue (http://qldspatial.information.qld.gov.au/catalogue/) under the Department of Natural Resources and Mines CC BY license. Bays were selected as representing an encapsulating body of water including some estuarine and tidal habitats, within which a population may usually reside irrespective of arbitrary coordinates. Standings were mapped using ArcGIS and then overlapped with the Bay layer. For embayment assessment, strandings were only used if they occurred within the defined bay area.

6.3.4. Model formation

When constructing the model, environmental discharge, air temperature and rain variables were lagged up to 12 months, with a cumulative effect. Time lag one included the environmental factor from time 0 and time -1, time lag two included the environmental factor from time 0, -1 and -2; and so on. A non-cumulative lag effect was also used for this analysis and compared against the cumulative effect.

A 12-month maximum lag time was used as there has historically been links made between marine turtle and dugong deaths occurring within this time frame of extreme weather events (Limpus et al., 2011; Meager and Limpus, 2014; Preen and Marsh, 1995). As seagrass loss after extreme weather events has been noted to be immediate it is not through to delay the response observed in marine turtle stranding rates (McKenzie et al., 2000).

All species of marine turtle to occur within the study area were analysed individually and collectively as a total count of strandings.

Age classes used for analysis were large immature, adult sized, small immatures, combined small immatures and large immatures, combined large immature and adult sized as well as all age classes together. Models were analysed where sample sized allowed.

The latitudes with the most strandings (both embayments and whole blocks) were chosen to run the models. These latitudes were -27°, -25°, -23°, -19°.

6.4. Model computation

The models were run as general linear models using R (R Core Team, 2016) with the bbmle package used to calculate additional information criterion including weights and qAIC values (Bolker, 2017; Bolker and R Development Core Team, 2016). The models were run *a priori* approach due to the complexity and number of possible models(Bolker et al., 2009; Burnham and Anderson, 2002; Doherty et al., 2012). Steps followed were similar to those outlined in Bolker et al (Bolker et al., 2009). Briefly these were specifying the effects, choosing an error distribution, graphically checking variance, fit GLM model to both full model and with each factor.

The strandings data used had an excess number of zeros, the data was also over-dispersed as such quasipoisson error distribution was used (Zeileis et al., 2008).

6.5. Model Hypothesis

The hypotheses tested are outlined below:

i) Small minimum air temperature will cause increases in marine turtle stranding rates.
ii) Maximum air temperature will not affect marine turtle stranding rates.
iii) Increased rainfall will cause increased marine turtle strandings rates 7-9 months later.

iv) Increased freshwater discharge will cause increased marine turtle stranding rates 7—9 months later.

v) All environmental factors combined will affect marine turtle stranding rates 7-9 months later.

6.5.1. Model testing

To begin with, models were run with all variables combined. These models proved non-significant ($p>0.1$). After this, each environmental factor was run separately to determine the individual effect. This was done for each age class and species for each latitude chosen. A no effect model was also run for each variable, ageclass and species.

In order to compare models, QAIC weights were calculated using the relative likelihood of the model. This was done following the steps outlined in Bolker (Bolker, 2017), briefly the regular model was fit, then the over dispersion parameter was manually extracted to calculate a qAIC value. qAIC is the quasi Akaike Information Criterion (AIC). qAIC weights allow for the selection of a "best approximating model" (Burnham and Anderson, 2002). This was then used in conjunction with the significance of the variables to determine which model most accurately explained the variance.

Each model was visually inspected to determine that both characteristics were met.

Strandings numbers of less than 10 over the 18-year period were excluded due to small sample size as were age class and species with less than 2 turtles per month for the 18-year period.

6.6. Results

6.6.1. Numbers of animals reported stranded

The number of turtles reported stranded over the 18 years is depicted in **Table 6.1.**

Table 6.1 Number of marine turtles reported stranded in each latitudinal block. NA represents not analysed. Bolded latitudes are the recognised hotspots.

Latitude	Number of strandings	
	Whole Block	Embayment
-28	102	NA
-27	**5344**	**1391**
-26	1302	NA
-25	**1572**	**410**
-24	642	NA
-23	**1256**	**158**
-22	228	NA
-21	**463**	**NA**
-20	496	NA
-19	**1390**	**417**
-18	282	NA
-17	237	NA
-16	411	NA
-15	65	NA
-14	26	NA
-13	1	NA
-12	19	NA
-11	10	NA
-10	7	NA

Upon initial investigation green turtles were the only species which could be analysed separately due to sample size. For the remaining sections of this study green turtles and the total number of strandings were analysed and reported.

6.6.2. Green turtles

6.6.2.1. *Rainfall*

Table 6.2 summarizes the relationship between rainfall and green turtle strandings rates. In brief it shows that within the -19° and -27° blocks strandings decreased as rainfall increased, while the -23 and -25° blocks showed split responses; the majority of age classes showed significant responses within the first 3 months; obvious differences between cumulative and non-cumulative effects of rainfall; different responses time noted with both embayments and whole blocks.

QAIC's for all groups assessed were different and no patterns were observed **(Table 6.2)**. In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.2 . Model results for green turtles and rainfall

↑ denotes increased strandings rates with increased rainfall. ↓ denotes decreased stranding rates with increased rainfall. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	2-8↓	12	1↓,2↓,4↓	12	-		-	12,11
	SI	0-12↓	12	1↓,10↓	12	0↓	12,11,10	-	12,11
	LI	1-8↓	12	4-,5↓	12,11,10	1↓,3-4↓	12,11,10	4↓	12,11
	A	-	12	-	12,11,10	0↑	12,11,10	-	12,11
	ALL IMM	0-12↓	12,11,10	1↓,2↓,4↓	12,11	0-1↓	12,11,10	-	12,11
	LARGE	4-5↓	12	4↓	12,11	3↓	12,11,10	-	12,10
-25	ALL	0-3↓,7-12↑	10,9,11	0-5↓,9-12↑	2,11,3,12	0-2↓,4-9↑,12↓	8	0-2↓,4-9↑,12↓	8,9
	SI	5-12↑	11,10	8-12↑	11,10,12	3-9↑	7,8	3-9↑	9,8
	LI	0-5↓,9-10↑	2,3,11	0-6↓	12,11,3	0-2↓,5-8↑,11-12↓	8,12	0-2↓,5-8↑,11-12↓	12,11
	A	0-5↓,8-12↑	2,1	0-6↓,9-11↑	2,0,1	0-2↓,5-9↑,12↓	8	0-2↓,5-9↑,12↓	9,7,8
	ALL IMM	0-2↓,6-12↑	10,9,11	0-5↓,9-12↑	11,12,10	0↓,4-9↑	7,8	0↓,4-9↑	8,9
	LARGE	0-5↓,8-12↑	2	0-6↓	2,3,1	0-2↓, 5-9↑,11-12↓	8	0-2↓, 5-9↑,11-12↓	8,9

Table 6.2 Continued

-23	ALL	7-12↑	12,11	0-2↓,7-12↑	12,11,10	6-11↑	12	0↑, 6-9↓	7,8
	SI	2↓,7-12↑	11,12,10	2↓,7-12↑	12,10,11,9	5-9↑	12,1	6-9↓	7,9,8
	LI	8-12↑	12,11,10,9	-	-	1↓,6-10↑	2,12	-	-
	A	0-2↑, 10-12↑	12	0	12,11,10	0↑, 10-12↑	10,12,11	0↓,8↑	11,10,12,9
	ALL IMM	2↓,7-12↑	11,12,10	0-2↓,7-12↑	12,11,10,9	5-9↑	12,11	0↓,4↑,6-9↑	7,9,8
	LARGE	0-1↑,8-12↑	12	0-1↓,11-12↑	12,11,10	0↑,8-11↑	12,10	1↓,8↑	0,11,9

-19	ALL	0-7↓,12↓	12	0-6↓,11-12↓	12,2,11	0↓,2↓	12	0↓,2↓	12,11,0,2
	SI	0-6↓	2,3,1,4	0-7↓	2,3,1,4	0-1↓	12,1	0-2↓	0,1,12
	LI	0-12↓	12,5,11	11-12↓	12,11,10	0↓,2↓	2,12	2↓,12↓	12,2,10,11
	A	11-12↓	12	-	12	10↓	10,12,11	10-11↓	10,11,12,9
	ALL IMM	0-7↓	3,2,4	0-7↓	2,3,4	0-2↓	12,11	0-2↓	0,2,11
	LARGE	1↓,5↓,10-12↓	12	11-12↓	12	2↓,10↓	12,10	10↓,12↓	12,10,11

6.6.2.2. *Cumulative Mean and Mean Freshwater Discharge*

Similar patterns in response for cumulative mean discharge and mean discharge and stranding rates were noted **(Table 6.3 and 6.4)**. There were different lag response times but the patterns remained the same. As such, analysis for both measures are discussed together. The only exception was all green turtles within the -25° block for cumulative lag effects of cumulative mean discharge in the whole block did not show the initial decrease that was observed in the mean discharge **(**Table 6.3 **and 6.4)**.

Differences in the examined latitudinal blocks were observed **(Table 6.3 and 6.4).**

Within each examined latitudinal block, there were no observed pattern as to which age class was the first to show significant responses **(Table 6.3 and 6.4)**.

These patterns did not change when comparing embayment's with whole blocks but the lag time may be extended when examining strandings within the embayment compared to whole block strandings **(Table 6.3 and 6.4)**.

All examined latitudinal blocks for non-cumulative lagged effects responded similarly to cumulative effects, with non-cumulative showing responses first **(Table 6.3 and 6.4)**.

QAIC's for all groups assessed were different and no patterns were observed **(Table 6.3 and 6.4)**. In most cases, the QAICs corresponded with significant model responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.3 Model results for green turtles and cumulative mean discharge.

↑ denotes increased strandings rates with increased discharge. ↓ denotes decreased stranding rates with increased discharge. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	7-12↑	12	8-12 ↑	12	1↑,6-9↑	8	1↑,5↑,8-10↑	12
	SI	5-12↑	12	7-12↑	12,11	6-9↑,11↑	7,8	6↑,8-11↑	8
	LI	11-12↓	12,11	12↓	12,11	-	11,12	-	11,12
	A	9-10↑	12,11	9-12↑	12,11	7-9↑	12,11.10	8-9↑	12,11
	ALL IMM	7-12↑	12	8-11↑	12,11	5-8↑	12,8,7	6↑,8-9↑	12,8
	LARGE	0↓	12	-	12,11	1↓,8↑	12,8	5↑	12,11
-25	ALL	6-12↑	10,11,9	3↓,8-12↑	11,10	5-9↑	8	6-10↑	8
	SI	4-12↑	11,10	7-12↑	11,12,10,9	3-10↑	7,8	6-11↑	8,7
	LI	0-4↓,8-12↑	12,11	1-4↓	11,12,10	0-2↓,5-9↑	8,7	1↓,6-9↑	12,11,8
	A	0-3↓,8-12↑	10,12,11	0-2↓,8-12↑	11,10,12	0-1↓,6-9↑	8	0↓,6-9↑	7,8
	ALL IMM	6-12↑	10,11	7-12↑	11,12,10	4-9↑	7	6-10↑	8,9
	LARGE	0-4↓,8-12↑	10,11,12	0-4↓,8-12↑	11,12,10	0-2↓,5-9↑	8	0↓,6-9↑	8,7

Table 6.3 Continued

-23	ALL	3-12↑	11,12	4-12↑	11,10,12	3-11↑	7	3-11↑	7
	SI	4-12↑	11,10	3-12↑	11,10,12,9	3-11↑	7	3-11↑	6,7
	LI	7-12↑	11,12,10,9	-	-	5-11↑	8,7	-	-
	A	0-12↑	12	7-12↑	11,10,12,9	0-1↑,6-12↑	10,9	6-8↑	7,8
	ALL IMM	4-12↑	11,10	3-12↑	11,10,12	3-11↑	7	3-11↑	6,7
	LARGE	0-12↑	12	7-12↑	11,12,10	0↑,5-12↑	8,10,9	7-11↑	8,7,10

-19	ALL	5-12↑	10,9,11	4-12↑	9,10,11	3-8↑	7,8,6,5	3-8↑	5,6
	SI	5-12↑	9,10	5-12↑	9,10,11,8	3-8↑	6,5,7	3-8↑	5,6
	LI	6-12↑	11,10,12,9	6-12↑	9,10,8,11	3↑,6-9↑	8,9	3-4↑,6↑	3,8,6,4
	A	-	11,12,10	4-12↑	10,11,9,8	3↑	3	3-5↑	3
	ALL IMM	5-12↑	10,9	5-12↑	9,10,11,8	3-9↑	6,7,5	3-8↑	5,6
	LARGE	8-12↑	11,10,12,9	4-12↑	10,9,8,11,12	3↑	3	3-5↑	3

Table 6.4. Model results for green turtles and mean discharge.

↑ denotes increased strandings rates with increased discharge. ↓ denotes decreased stranding rates with increased discharge. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	7-1 2↑	12	8-12↑	12	1↑,6-9↑	8	1↑,5↑,8-10↑	12
	SI	5-12↑	12	7-12↑	12,11	6-9 ↑,11↑	8	6↑,8-11↑	8
	LI	11-12↓	12,11	12↓	12,11	-	12,11	-	11,12
	A	9-10↑	12,11	9-12↑	12,11	7-9↑	12	8-9↑	12,11
	ALL IMM	7-12↑	12	8-12↑	12,11	5-8↑	12,8	6↑,8-9↑	12,8
	LARGE	0↓	12	-	12	1↓,8-9↑	12,8	5↑	12,11
-25	ALL	2↓,7-12↑	10,11	2-3↓,8-12↑	11,10	5-9↑	8	6-10↑	8
	SI	5-12↑	11,10	7-12↑	11,12,10,9	3-10↑	7,8	6-11↑	8
	LI	0-4↓,8-11↑	12,11	1-4↓	11,12,10	0-2↓,5-9↑	8,7	1↓,6-9↑	12,11
	A	0-3↓,8-12↑	10,12,11	0-2↓,8-12↑	11,10,12	0-1↓,6-9↑	8	0↓,7-9↑	7,8,9
	ALL IMM	6-12↑	10,11	7-12↑	11,12,10	4-9↑	7	6-10↑	8,9
	LARGE	0-4↓,8-12↑	10,11,12	0-4↓,8-12↑	11,12,10	0-2↓,5-9↑	8	0↓,6-9↑	8,7

Table 6.4 Continued

-23	ALL	3-12↑	11,12	4-12↑	11,10,12	3-11↑	7	3-11↑	7
	SI	4-12↑	11,10	3-12↑	11,10,12,9	3-11↑	7	3-11↑	6,7
	LI	7-12↑	11,12,10,9	-	-	5-11↑	8,7	-	-
	A	0-12↑	12	7-12↑	11,10,12,9,8	0-1↑,6-12↑	10,9	7-8↑	7,8
	ALL IMM	4-12↑	11,10	3-12↑	11,10,12	3-11↑	7	3-11↑	6,7
	LARGE	0-12↑	12	7-12↑	11,12,10	0↑,5-12↑	8,10	7-11↑	8,7,10

-19	ALL	5-12↑	10,9,11	4-12↑	9,10,11	3-8↑	7,8,6,5	3-8↑	5,6
	SI	5-12↑	9,10	5-12↑	9,10,11,8	3-8↑	6,5,7	3-8↑	5,6
	LI	6-12↑	11,10,12,9	6-11↑	9,10,8,11	3-4↑,6-9↑	8,9	3-4↑,6↑	3,8,6,4
	A	-	11,12,10,9	4-12↑	10,11,9,8	3↑	3	3-5↑	3
	ALL IMM	5-12↑	10,9	5-12↑	9,10,11,8	3-9↑	6,7,5	3-8↑	5,6
	LARGE	8-12↑	11,10,12,9	4-12↑	10,9,8,11	3↑	3	3-5↑	3

6.6.2.3. *Peak discharge*

Table 6.5 summarizes the relationships between green turtle stranding numbers and peak discharge. In brief, large immatures and large turtles in the -27° block showed no significant response; in the -19°, -23° and -27° degree blocks, as peak discharge increased so did green turtle stranding rates, as a comparison, in the -25° block showed a split response with strandings decreasing with increased discharge over the first 5 months, which then switched to increased strandings with increasing peak discharge.

Table 6.5 also displays that within each examined latitudinal block, most age classes showed a significant stranding response to peak discharge, however, there was no observed pattern as to which age class was the first to show significant responses; all examined latitudinal blocks for non-cumulative lagged effects of peak discharge, responded similarly to cumulative effects, with non-cumulative showing responses first.

These patterns did not change when comparing embayments with whole blocks but the lag time may be extended when examining strandings within the embayment compared to whole block green turtle strandings **(Table 6.5)**.

QAIC's for all groups assessed were different and no patterns were observed **(Table 6.5)**. In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.5 Model results for green turtles and peak discharge

↑ denotes increased strandings rates with increased discharge. ↓ denotes decreased stranding rates with increased discharge. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	7-12↑	12	9-12↑	12	6-9↑	8	8-10↑	12,10
	SI	6-12↑	12	8-12↑	12,11	6-9↑,11↑	11,8,12	6↑,8-11↑	8
	LI	-	12,11	-	12,11	-	11,12	-	12,11
	A	9-12↑	12,11	8-12↑	12,11	9-10↑	12,11,10	8-9↑	12,11
	ALL IMM	7-12↑	12	10-12↑	12,11	6-8↑	12	8↑,10↑	12,10
	LARGE	-	12	-	12	8↑	12,9,8	-	12,11
-25	ALL	2↓,7-12↑	10,11	3↓,8-12↑	11,10	6-9↑	8	6-10↑	8
	SI	5-12↑	11,10	7-12↑	11,12,10,9	5-10↑	7	6-10↑	8,7
	LI	1-5↓,8-12↑	11,12	1-4↓,9-11↑	11,12,10	1-2↓,5-9↑	8	1↓,3↓,5↑,7-9↑	12,11,8
	A	0-5↓,8-12↑	10,11,12	0↓,9-12↑	11,10,12	0-1↓,6-9↑	8	0↓,7-9↑	8,9,7
	ALL IMM	6-12↑	10,11	8-12↑	11,12,10	5-9↑	7,8	7-9	8
	LARGE	0-5↓,8-12↑	10,11,12	0-5↓,9-12↑	11,12,10	0-2↓,6-9↑	8	0↓,7-9↑	8

Table 6.5 Continued

-23	ALL	4-12↑	11,12	5-12↑	11,10,12	3-11↑	7	3-11↑	7
	SI	4-12↑	11,10	4-12↑	11,10,12,9	3-11↑	7	3-11↑	6,7
	LI	6-12↑	11,12,10,9	-	-	3↑,5-11↑	7,8	-	-
	A	0-12↑	12	7-12↑	11,10,12,9	0-1↑,6-12↑	12,10,9,8	6-8↑,10↑	7,8,10
	ALL IMM	4-12↑	11,10	4-12↑	11,10,12,9	3-11↑	7	3-11↑	6,7
	LARGE	0-12↑	12	7-12↑	11,12,10	0↑,5-12↑	8,7	6-11↑	7,8,10

-19	ALL	5-12↑	10,9	4-12↑	9,10	3-8↑	3	3-8↑	3,5
	SI	5-12↑	9,10	5-12↑	9,10,11	4-8↑	5,6	4-8↑	5
	LI	6-12↑	10,11,9	5-12↑	9,8,10,11	3-4↑,6↑, 8-9↑	8,3,9	3-4↑,6↑	3,6,4,8
	A	4-11↑	10,11,9,12	3-12↑	10,9,8	3↑	3	3-5↑	3
	ALL IMM	5-12↑	10,9	5-12↑	10,9,11	3-9↑	6,5,7	3-8↑	5,6
	LARGE	8412↑	10,11,9	3-12↑	9,10,8,	3↑	3	3-5↑	3

6.6.2.4. *Monthly mean maximum air temperature*

Table 6.6 summarizes the relationship between monthly mean maximum air temperature and green turtle stranding rates. In brief it shows that in most cases, as monthly mean maximum air temperatures increased the green turtle stranding rate decreased; there was a significant response noted within the first 4 months; there were very obvious differences between cumulative and non-cumulative effects of monthly mean maximum air temperature, with non-cumulative effects more likely to produce split responses; and that there were similar stranding response times noted with both embayments and the whole blocks for monthly mean maximum air temperature.

QAIC's for all groups assessed were different and no patterns were observed **(Table 6.6)**. In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.6. Model results for green turtles and monthly mean maximum air temperature.

↑ denotes increased strandings rates with increased monthly mean maximum air temperature. ↓ denotes decreased stranding rates with increased monthly mean maximum air temperature. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	0-9 ↓	6	0-9↓	6	0-4↓,6-10↑,12↓	8,9	0-4↓,6-10↑,12↓	8
	SI	0-8↓,12 ↓	4,12	0-9↓,11-12↓	12,6	0-4↓,6-9↑,12↓	12,8	0-4↓,6-10↑,12↓	8
	LI	1-8 ↓	6	2-8↓	12,11	1-5↓,7-11↑	9	1-4↓,7-10↑	10,8,12
	A	1-9 ↓	6	0-7↓	12,11	1-5↓,7-11↑	9	0-3↓,6-9↑	12,8
	ALL IMM	0-9 ↓	5	0-9↓,12↓	7,6	0-4↓,6-10↑	8	0-4↓,6-10↑,12↓	8,7
	LARGE	1-9 ↓	6,5	0-8↓	12,11	1-5↓,7-11↑	9	0-4↓,6-10↑	8,9
-25	ALL	1-8↓,11↑	5,4	2-8↓,10-12↑	11,5,12,6	0-4↓, 6-10↑	8	1-5↓, 7-11↑	9
	SI	0-5↓,10-11↑	10,3,11	3-7↓,11↑	11,12,5	0-3↓, 5-10↑,12↓	7	2-4↓,7-10↑	9,10,8
	LI	2-8↓	5,4	2-8↓,11-12↑	11,12	1-4↓, 6-10↑	3,9,8	1-4↓, 7-11↑	9,11,10
	A	2-9↓	6,5	0↑,3-8↓,11-12↑	11,12,5	0-5↓, 7-11↑	9	0↑,2-5↓, 7-11↑	9,8
	ALL IMM	0-7↓,10-11↑	4,3	2-8↓,11-12↑	11,12,5,6	0-3↓, 6-10↑	7,8	1-4↓, 7-11↑	9,8
	LARGE	2-8↓	5,6	2-8↑↓,11-12↑	11,12,5,6	1-5↓, 7-11↑	3,9	1-5↓, 7-11↑	9

Table 6.6 Continued

-23	ALL	0-12↓	6,5,7	0-12↓	4,3,12,5	1-4↓, 8-9↑	3,2	0-3↓, 6-8↑,12↓	2,3,1,12
	SI	0-7↓,11-12↓	3,4	0-12↓	4,12,3,5	0-3↓, 6-8↑,12↓	1,2,12	0-3↓, 7-8↑,12↓	2,1,3
	LI	1-11↓	5,4,6	-	-	1-4↓,7-9↑	2,3,8,12	-	-
	A	0↑↓,5-12↓	9,8,10	-	12,11,10	0↑, 3-7↓,9-12↑	5	-	12,11,10
	ALL IMM	0-8↓, 10-12↓	3,4	0-12↓	4,3,5,12	0-3↓, 6-8↑,12↓	2,1	0-3↓, 7-8↑,12↓	2,3,1,12
	LARGE	4-12↓	8,9	2-5↓	12,11,10	2-6↓,9-11↑	5,4	1-3↓	12,2,11,10

-19	ALL	0-12↓	12	0-12↓	12	0-3↓, 6-7↑,11-12↓	12	0-2↓, 6-7↑,11-12↓	12
	SI	0-12↓	12	0-12↓	12	0-3↓, 6-8↑,11-12↓	1	0-3↓, 6-7↑,11-12↓	1
	LI	0-12↓	12	0-6↓,10-12↓	12	0-3↓,12↓	12	0-3↓,7↑,12↓	12,1,0,11
	A	0-3↓,11-12↓	12	0-4↓,11-12↓	12	0-1↓, 11-12↓	12	0-1↓, 6↑,11-12↓	12
	ALL IMM	0-12↓	12	0-12↓	12	0-3↓, 6-8↑,11-12↓	1	0-3↓, 6-7↑,11-12↓	1,12
	LARGE	0-5↓,11-12↓	12	0-5↓,10-12↓	12	0-1↓,12↓	12	0-1↓,6↑,12↓	12

6.6.2.5. *Monthly mean minimum air temperature*

Table 6.7 summarizes the relationship between monthly mean minimum air temperature and green turtle stranding rates. In brief it shows in most cases, as monthly mean minimum air temperatures increased the stranding rate decreased; there were very obvious differences between cumulative and non-cumulative effects, with non-cumulative effects resulted in split responses; in most cases there was a significant green turtle strandings response noted within the first 3 months of the mean minimum air temperature recorded; there were similar responses time noted with both embayments and whole blocks.

QAIC's for all groups assessed were different and no patterns were observed **(Table 6.7)**. In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.7 Model results for green turtles and monthly mean minimum air temperature

↑ denotes increased strandings rates with increased monthly mean minimum air temperature. ↓ denotes decreased stranding rates with increased monthly mean minimum air temperature. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	0-9 ↓	6	0-9↓	6	0-4↓,6-11↑,12↓	8	0-4↓,6-10↑,12↓	8,9
	SI	0-8↓,12 ↓	4,12	0-8↓	7,6	0-3↓,6-10↑,12↓	8	0-4↓,6-10↑,12↓	8,9
	LI	1-9 ↓	6	1-8↓	12,11	1-4↓,7-11↑	9	0-4↓,8-9↑	9,12,10
	A	1-9 ↓	6	0-6↓	12,11	1-5↓,7-11↑	9	0-3↓,5-9↑,11-12↓	12
	ALL IMM	0-9↓,12 ↓	4,5	0-8↓	7,6	0-4↓,6-10↑,12↓	8	0-4↓,6-10↑,12↓	8,9
	LARGE	1-9 ↓	6	0-8↓	12,11	1-5↑,7-11↓	9	0-4↓,6-9↑,12↓	9,8
-25	ALL	0-8↓,12↓	4,5	1-10↓	5,6,4	0-4↓, 6-10↑, 12↓	8,2	1-5↓,7-11↑	9,2
	SI	0-6↓,12↓	12,2,3	1-10↓,12↓	5,6,4,7	0-3↓, 5-8↑,11-12↓	12	1-4↓, 7-10↑	9,2
	LI	1-8↓	4,5	1-9↓	12,7,11	1-4↓, 6-10↑,12↓	8,9	1-4↓, 6-10↑	9,8
	A	1-9↓	5,6	1-10↓	5,6,7,4	1-5↓, 7-11↑	9	1-5↓, 6-10↑	9,8
	ALL IMM	0-7↓,102↓	3,4	1-10↓,12↓	5,4,6,7	0-3↓, 6-10↑,12↓	7,2	1-4↓, 6-10↑	9,8
	LARGE	1-9↓	5	1-10↓	5,6,4,7	1-4↓,6-11↑	9	1-5↓,7-11↑	9,8

Table 6.7 Continued

-23	ALL	0-12↓	5,4,6	0-8↓,10-12↓	12,3,2,4	0-4↓, 7-9↑	2	0-3↓, 6-8↑,12↓	2,1,7
	SI	0-7↓,11-12↓	3,2	0-7↓,10-12↓	3,2,4,12	0-3↓, 6-8↑,12↓	1	0-3↓, 6-9↑,12↓	2,1,7
	LI	0-12↓	4,12,5,6	-	-	0-4↓,6-9↑,12↓	2,8,12	-	-
	A	0↑,5-10↓	8,9	12↓	12,11,10	0↑, 3-6↓,9-12↑	11,5,10	2↓	12,11,10
	ALL IMM	0-7↓,11-12↓	3,2	0-8↓, 10-12↓	3,2,4,12	0-3↓, 6-9↑,12↓	1,2	0-3↓, 6-9↑	2,1,7
	LARGE	3-11↓	8,9,7	1-5↓,11-12↓	12,11,10	2-6↓,8-112↑	10,11	1-3↓	12,2,11,10

-19	ALL	0-5↓,12↓	12	0-5↓,12↓	12	0-2↓, 6-8↑,11-12↓	12	0-2↓, 5-8↑,11-12↓	12,0
	SI	0-6↓,12↓	2,3	0-6↓,12↓	2,1,3	0-3↓, 5-9↑,11-12↓	1,0,7	0-2↓, 5-8↑,11-12↓	0,1,7
	LI	0-6↓,11-12↓	12	0-4↓,12↓	12	0-3↓,7-8↑,12↓	12,8,2	0-2↓,6-7↑,12↓	12,7,1
	A	0-3↓,11-12↓	12	0-2↓,12↓	12	0↓, 11-12↓	12	0-1↓, 5-6↑,11-12↓	12,11,0
	ALL IMM	0-6↓,12↓	2 ,3	0-5↓,12↓	2,1,12,3	0-3↓, 5-9↑,12↓	1,7	0-3↓, 5-8↑,11-12↓	0,1,12
	LARGE	0-3↓,11-12↓	12	0-3↓,12↓	12	0-1↓,12↓	12	0-1↓,5-7↑,11-12↓	12

6.6.2.6. *Monthly average daily diurnal air temperature difference*

Table 6.8 summarizes the relationship between monthly average daily diurnal air temperature difference and green turtle stranding rates. In brief it shows very obvious differences between cumulative and non-cumulative effects of monthly average daily diurnal air temperature differences; Non-cumulative effects resulted in split responses whereas in most cases the cumulative effects resulted in decreased stranding rate with increased mean minimum air temperature; in most cases there was a significant response noted within the first 3 months of monthly average daily diurnal air temperature difference being recorded; similar response times were noted with both embayments and whole blocks.

The exception was the -19° block, where significant response times were varied for small immatures, immature and all green turtles and adults and large turtles within the -19° block did not display a significant response **(Table 6.8)**.

QAIC's for all groups assessed were different and no patterns were observed. In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce significant relationship **(Table 6.8)**.

Table 6.8 Model results for green turtles and monthly average daily diurnal air temperature difference.

↑ denotes increased strandings rates with increased average daily diurnal air temperature difference. ↓ denotes decreased stranding rates with increased average daily diurnal air temperature difference. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum - Whole		Cum - bay		Non-Cum Whole		Non-Cum Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	0-8 ↑	6	0-8↑	12	0-4↑,6-10↓	8	0-3↑,6-10↓,12↑	8,9
	SI	0-6 ↑	12,11	0-6↑	12,11	0-3↑,6-10↓,12↑	8	0-3↑,6-10↓	8,9
	LI	1-9 ↑	6	0-8↑	12,11,10	1-4↑,7-11↓	9	0-3↑,8-9↓	9,12,11
	A	1-8 ↑	5,4	0-5↑	12,11	1-4↑,7-10↓	9	0-2↑,5-9↓,11-12↑	12
	ALL IMM	0-7 ↑	4	0-7↑	12,11	0-4↑,6-10↓,12↑	8	0-3↑,6-10↓	9,8
	LARGE	1-8 ↑	6	0-8↑	12,11	1-4↑,7-11↓	9	0-3↑,6-9↓,12↑	12
-25	ALL	0-8↑,12↑	5,4	0-12↑	5,4,3,2	0-3↑, 6-9↓,12↑	2,1	0-4↑, 7-10↓	2
	SI	0-7↑,11-12↑	10,3,11	0-12↑	4,5,3,2	0-2↑,6-8↓,11-12↑	12	0-4↑,9↓,12↑	12,0
	LI	0-7↑	5,4	0-9↑,12↑	12,11,4	0-3↑,6-9↓,12↑	8,7	0-3↑,6-9↓,12↑	12,9
	A	0-8↑	6,5	0-12↑	5,4,6,3	0-4↑, 6-10↓,12↑	8,9,2	0-4↑, 7-10↓	2
	ALL IMM	0-7↑,12↑	4,3	0-12↑	4,3,5,2	0-3↑, 5-9↓,11-12↑	12,0	0-4↑, 6-10↓,12↑	12,1
	LARGE	0-8↓	5,6	0-12↑	5,4,3	0-4↑,6-10↓,12↑	8	0-4↑,7-10↓,12↑	2

Table 6.8 Continued

-23	ALL	0-3↑	12,11,10	0-3↑,	12,11,2,10	0↑, 5-9↓	9,8	0↑, 5-7↓	5,6,12,9
	SI	0-2↑	12,11,10,9	0-1↑,8-12↓	12,11,10,9	0↑,5↓	5	0↑,5-7↓,9↓	5,9,7,6
	LI	0-4↑	12,11,10	-	-	0-1↑,6↓	12,11,10	-	-
	A	11-12↓	12	2-4↑	12,11,10	3-4↑,6↓,8-11↓	10,9,12,8	2↑	2,11,12,10
	ALL IMM	0-3↑	12,2,11	0-2↑, 9-12↓	12,11,10	0↑, 5↓	5	0↑, 5-7↓,9↓	6,7,5
	LARGE	3-4↑	12	2-4↑	12,11,10	6↓,9-10↓	10,9,12	2↑	11,12,10,2

-19	ALL	0-3↑, 8-12↓	10	0-3↑, 7-12↓	10,9	0-2↑, 5-8↓	7,8,6	0↑, 4-8↓	7,8,6
	SI	0-3↑,7-12↓	10,11	0-3↑,7-12↓	10,11,9	0-2↑,4-9↓	7,6	0-1↑,5-9↓	7,6
	LI	-	11,12,10	8-10↓	9,10,11,8	6↓,8↓	8,6	5-7↓	6,7,5,8
	A	-	12,9,8	7-10↓	9,8,10	-	12,11	3-5↓	5,11,7,12
	ALL IMM	0-3↑, 7-12↓	10,11	0-3↑, 7-12↓	10,11,9	0-2↑, 5-9↓	6,7	0-2↑, 4-9↓	7,6,8
	LARGE	-	10,9,11	7-10↓	9,8,10	-	12,8,7,11	4-7↓,11↑	5,11

6.6.3. All Marine Turtle Strandings

6.6.3.1. *Rainfall*

Table 6.9 summarizes the relationship between rainfall and marine turtle stranding rates. In brief it shows that when comparing rainfall across all blocks, there were different patterns noted for each block; cumulative effects within the -27° block stranding rates decreased as rainfall increased; non-cumulative effects within the -27° block showed mixed results; within each examined latitudinal block, there were similar stranding response times noted for embayments and the whole blocks; there were very obvious differences between cumulative and non-cumulative effects of rainfall on all turtle stranding rates (**Table 6.9**).

QAIC's for all groups assessed were different and no patterns were observed (**Table 6.9**). In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.9. Model results for all turtles and rainfall.

↑ denotes increased strandings rates with increased rainfall. ↓ denotes decreased stranding rates with increased rainfall. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	4↓	12,11	-	12,11,10	-	12,11	-	12,11
	SI	0-6↓	12,11	0-4↓	12,11,10	0-4↓	12,11	0	12,11,10
	LI	1-7↓	12,11,10	4↓	12,11,10	1,3-4	12,11,10	5↑,8↑	12,11,10,
	A	1↓	12,11	-	12,11	1	12,11,10	5↑	12,11
	ALL IMM	1-8↓	12,11	0-4↓	12,11,10	0-1	12,11	0↓	12,11,10
	LARGE	-	12,11	2↓	12,11	-	12,11,10	5↑	12,11,10
-25	ALL	0-4↓,7-12↑	10,11	0-5↓,9-12↑	2,11,3,12	0-2↓,4-9↑,12↓	8	0-2↓,6-9↑	8
	SI	5-12↑	11,10	8-12↑	11,10,12,9	3-9↓	7,8	0↓,5-10↑	8,7,9
	LI	0-5↓	2,3	0-6↓	12,11,3,2	0-2↓,5-8↑,11-12↓	12	0-3↓,7-8↑,12↓	12,11,8
	A	0-5↓,9-12↑	2,1,12	0-6↓,9-11↑	2,0,1,3	0-2↓,6-9↑,12↓	8,7	0-2↓,6-9↑	7,9,8
	ALL IMM	0-2↓,7-12↑	10,11,9	0-5↓,9-12↑	11,12,10,9	0↓,2↓,4-9↑	8,7	0-2↓,5-9↑,12↓	8
	LARGE	0-5↓,9-11↑	2	0-6↓	2,3,1	0-2↓,5-9↑,12↓	8	0-2↓,6-9↑,12↓	8,7,9

Table 6.9 Continued

-23	ALL	7-12↑	12,11	0-2↓,7-12↑	12,11,10	6-10↑	8,9	0↓,6-9↑	7,0,8
	SI	0-3↓,7-12↑	11,12,10	0↓,2↓,7-12↑	12,11,10	5-9↑	8,7	0↓,4↑,6-9↑	7
	LI	-	-	-	-	-	-	-	-
	A	0-2↑,10-12↑	12	0-1↓	12,11,10	9-12↑	11,10,12	0↓,8↑	11,10,12,9
	ALL IMM	0-3↓,7-12↑	11,12,10	0-2↓,7-12↑	12,11,10	5-9↑	8	0↓,4↑,6-9↑	7
	LARGE	0↑,8-12↑	12	0-2↓,11-12↑	12,11,10	8-11↑	10,11	0↓,8↑	0,11,10,9
-19	ALL	0-7↓,12↓	12	0-6↓,12↓	12,2,11	0↓,2↓	12	0↓,2↓	12
	SI	0-6↓	2,3,4	0-7↓	2,1,3,4,5	0-1↓	12,11	0-1↓	0,1
	LI	0↓,2-7↓,10-12↓	12,5,11,4	-	12,11	0↓,2↓	12,10	12↓	12,10,11
	A	11-12↓	12	11-12↓	12,11	10↓	10,12,11	10↓	10,12,11
	ALL IMM	0-7↓	4,3,2	0-7↓	2,3,12	0-2↓	12,12,10	0-2↓	10,12,11
	LARGE	5↓,11-12↓	12	12↓	12,11	2↓		10↓,12↓	12,0,11

6.6.3.2. *Cumulative Mean and Mean Discharge*

Table 6.10 and 6.11 summarizes the relationship between cumulative mean, mean discharge and all marine turtle stranding rates. In brief it shows in most cases, as cumulative mean discharge increased, the stranding rate for all turtles also increased; each examined latitudinal block, there were similar stranding response times noted for embayments and the whole blocks in respect to discharge; within each examined latitudinal block, there were also similar response times for cumulative effect vs non-cumulative effect of discharge.

The exceptions for this patterns were the -25° block which showed a split response. The small immatures and all immature turtles within the -25° block did not show a split response, instead showed increased strandings with increasing discharge; the -19° and -23° blocks showed very similar response times to each other. The -25° and -27° blocks showed similar responses to each other (**Table 6.10 and 6.11)**.

QAIC's for all groups assessed were different and no patterns were observed (**Table 6.10 and 6.11**). In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.10. Model results for all turtles and cumulative mean discharge.

↑ denotes increased strandings rates with increased discharge. ↓ denotes decreased stranding rates with increased discharge. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	8-12↑	12,11	9-12↑	12	1↑,7-9↑	8	1↑,7-10↑	8,10
	SI	5-12↑	12,11	6-12↑	12,11	6-12↑	7,11,8	1↑,5-11↑	8,11
	LI	-	12,11,10	-	12,11,10	-	12,11,10	-	12,11,10
	A	8-12↑	12,11,10	8-12↑	12,11,10	7-9↑	8	10-Jul	8,9
	ALL IMM	7-12↑	12,11	8-12↑	12,11,10	6-11↑	7,8,11	1↑,7-11↑	11,8,12,10
	LARGE	9-11↑	12,11,10	9-12↑	12,11,10	7-9↑	8	7-10↑	12,10
-25	ALL	2↓,7-12↑	10,11	3↓,8-12↑	11,10,12	5-10↑	8	6-10↑	8
	SI	4-12↑	11,10,12	7-12↑	11,12,10	3-10↑	8,7	5-11↑	8,7,10,9
	LI	0-4↓,8-11↑	12,11,10	1-4↓	11,12,10	0-2↓,5-9↑	8,7,12	1↓,3↓,7-9↑	11,12,8
	A	0-3↓,8-12↑	12,10,11	0-2↓,8-12↑	11,10,12	0-1↓,6-9↑	8	0↓,6-10↑	7,8,9
	ALL IMM	6-12↑	10,11	7-12↑	11,12,10	4-10↑	7,8	5-10↑	8
	LARGE	0-4↓,8-12↑	12,11,10	0-4↓,8-12↑	11,12,10	0-2↓,5-9↑	8	0-1↓,6-9↑	8,7

Table 6.10 Continued

-23	ALL	4-12↑	11,12	3-12↑	11,10,12	3-11↑	7	3-11↑	7,6
	SI	4-12↑	11,10	3-12↑	10,11,9,12	3-11↑	7	3-11↑	6,7
	LI	-	-	-	-	-	-	-	-
	A	0-12↑	12	7-12↑	11,12,10	0-1↑,6-12↑	10,9	7-11↑	7,8,10
	ALL IMM	4-12↑	11,10	4-12↑	11,10,12	3-11↑	7	3-11↑	6,8
	LARGE	0-12↑	12	7-12↑	11,12,10	0↑,5-12↑	10,8,9	6-11↑	8,7,10
-19	ALL	5-12↑	10,11	4-12↑	10,9,11	3-9↑	7,8	3-8↑	5
	SI	5-12↑	10,9,11	5-12↑	10,9,11,8	3-9↑	6,7,5	3-8↑	5,6
	LI	6-12↑	10,11,9,12	4-12↑	9,10,8,11	3-4↑,6↑,8-9↑	8,9	3-4↑	4,3,8,6
	A	-	11,12,10	4-12↑	10,11,9,8	3↑	3	3-5↑	3,5
	ALL IMM	5-12↑	10,9	4-12↑	9,10,11,8	3-9↑	6,7,8	3-8↑	5,6
	LARGE	8-12↑	11,10,12	4-12↑	10,9,11,8	3↑	3	3-5↑	3,5,4

Table 6.11. Model results for all turtles and mean discharge.

↑ denotes increased strandings rates with increased discharge. ↓ denotes decreased stranding rates with increased discharge. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	8-12↑	12,11	9-12↑	12,11	1↑,7-9↑	8	1−,7-10↑	8
	SI	5-12↑	12,11	6-12↑	12,11	6-12↑	7,11,8,9	1↑,5-11↑	8,11
	LI	-	12,11,10	-	12,11,10	-	12,11,10	-	12,11,10
	A	8-12↑	12,11,10	8-12↑	12,11,10	7-9↑	8	7-10↑	8,9
	ALL IMM	7-12↑	12,11	8-12↑	12,11,10	6-11↑	7,8,11	1↑,7-11↑	11,18,12,10
	LARGE	9-11↑	12,11	9-12↑	12,11,10	7-9↑	8,12	7-10↑	12,10
-25	ALL	2↓,7-12↑	10,11	2-3↓,8-12↑	11,10,12	5-10↑	8	6-10↑	8
	SI	5-12↑	11,10,12	7-12↑	11,12,10,9	4-10↑	7,8	5-11↑	8,7,10
	LI	0-4↓,8-11↑	12,11,10	1-4↓	11,12,10	0-2↓,5-8↑	8,7,12	1↓,3↓,7-9↑	11,12,8
	A	0-3↓,8-12↑	12,10,11	0-2↓,8-12↑	11,10,12	0-1↓,6-9↑	8	0↓,6-10↑	7,8,9
	ALL IMM	6-12↑	10,11	7-12↑	11,12,10	4-10↑	7,8	5-10↑	8
	LARGE	0-4↓,8-12↑	12,11,10	0-4↓,8-12↑	11,12,10	0-2↓,5-9↑	8	0-1↓,6-9↑	8,7

Table 6.11 Continued

-23	ALL	4-12↑	11,12	4-12↑	11,10,12	3-11↑	7	3-11↑	7,6
	SI	4-12↑	11,10	3-12↑	10,11,9	3-11↑	7	3-11↑	6,7
	LI	-	-	-	-	-	-	-	-
	A	0-12↑	12	7-12↑	11,12,10	0-1↑,6-12↑	10,9	7-11↑	7,8,10
	ALL IMM	4-12↑	11,10	4-12↑	11,10,12	3-11↑	7	3-11↑	6,7
	LARGE	0-12↑	12	7-12↑	11,12,10	0↑,5-12↑	10,8,9	6-12↑	8,7,10,11
-19	ALL	5-12↑	10,11	4-12↑	10,9	3-9↑	7,8	3-8↑	5
	SI	5-12↑	10,9,11	4-12↑	10,9,11,8	3-9↑	6,7,5	3-8↑	5,6
	LI	6-12↑	10,11,9	4-12↑	9,10,8,11	3-4↑,6↑,8-9↑	8,9	3-5↑	4,3,8,6
	A	-	11,12,10	4-12↑	10,11,9,8	3↑	3,	3-5↑	3,5
	ALL IMM	5-12↑	10,9	4-12↑	9,10,11,8	3-9↑	6,7,8	3-8↑	5,6
	LARGE	7-12↑	11,10,12	4-12↑	10,9,11,8	3↑	3	3-5↑	3,5

6.6.3.3. *Peak Discharge*

Table 6.12 summarizes the relationship between peak discharge and marine turtle stranding rates. In brief it shows that in most cases, as peak discharge increased, the stranding rate also increased; each examined latitudinal block, there were similar stranding response times noted for embayments and the whole blocks for peak discharge; within each examined latitudinal block, similar response times for cumulative effect vs non-cumulative effect of peak discharge were observed.

The exceptions to these patterns were that the -25° block which showed a split response; small immature and all immature within the -25° block did not show a split response, instead showed increased strandings with increased discharge; large immatures within the whole -27° block showed a split response for cumulative effects and did not return significant responses for the non-cumulative effects or cumulative effects within the embayment; The -23° and -19° blocks showed very similar response times to each other (**Table 6.12**).

QAIC's for all groups assessed were different and no patterns were observed (**Table 6.12**). In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.12 Model results for all turtles peak discharge

↑ denotes increased strandings rates with increased discharge. ↓ denotes decreased stranding rates with increased discharge. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	8-12↑	12,11	8-12↑	12,11	7-10↑	8,7	1↑,8-10↑	8,10,12
	SI	5-12↑	12,11	8-12↑	12,11	6-12↑	11,9	1↑,6↑,8-11↑	11,8,10
	LI	3-4↓,6↓	12,11,10	-	12,11,10	-	12,11,10	-	12,11,10
	A	7-12↑	12,11,10	8-12↑	12,11,10	7-9↑	8	7-10↑	8,9
	ALL IMM	7-12↑	12,11	8-12↑	12,11,10	7-11↑	1,7,12	8-11↑	11,10,12
	LARGE	9-12↑	12,11	9-12↑	12,11,10	7-9↑	8,12,11	7-10↑	12,10
-25	ALL	1-2↓,7-12↑	10,11	3↓,8-12↑	11,10,12	6-9↑	8,7	6-10↑	8
	SI	5-12↑	10,11	7-12↑	11,10,12,9	6-10↑	7,5	6-10↑	8,7,10
	LI	0-5↓,8-12↑	11,12,10	1-4↓,9-11↑	11,12,10	0-2↓,5-9↑	8	1↓,3↓,7-8↑	12,8,11
	A	0-5↓,8-12↑	12,11,10	0↓,9-12↑	10,11,12,9	0-1↓,7-9↑	8	7-10↑	8,7,9
	ALL IMM	7-12↑	10,11	8-12↑	11,12,10	5-10↑	7,8	6-10↑	8
	LARGE	0-5↓,8-12↑	12,11,10	0-5↓,9-12↑	11,12,10	0-2↓,6-9↑	8	1↓,7-9↑	8,7,12

Table 6.12 Continued

-23	ALL	4-12↑	11,12	5-12↑	11,10,12	3-11↑	7	3-11↑	7,6
	SI	5-12↑	11,10	4-12↑	10,11,9,12	3-11↑	7	3-11↑	6,7
	LI	-	-	-	-	-	-	-	-
	A	0-12↑	12	8-12↑	11,12,10	0-1↑,6-12↑	10,9	7-11↑	7,10,8
	ALL IMM	5-12↑	11,10	4-12↑	11,10,12	3-11↑	7	3-11↑	6,7
	LARGE	0-12↑	12	8-12↑	11,12,10	0↑,5-12↑	8,10,7,9	6-11↑	10,7,8,11
-19	ALL	5-12↑	10,11	4-12↑	10,9	3-9↑	7	3-8↑	5
	SI	5-12↑	10,9	5-12↑	9,10,11,8	3-8↑	7,5,6	3-8↑	5
	LI	5-12↑	10,11,9,12	4-12↑	9,10,8	3-4↑,6↑,8-9↑	8,9	3-6↑	3,6,5,4
	A	5↑,7-12↑	11,10,12	3-12↑	10,9,11,8	3↑	3	3-5↑	3
	ALL IMM	5-12↑	10,9	5-12↑	9,10	3-9↑	6,7	3-8↑	5
	LARGE	5-12↑	11,10,9	3-12↑	9,10	3↑	3	3-5↑	3

6.6.3.4. *Monthly Mean Maximum Air Temperature*

Table 6.13 summarizes the relationship between monthly mean maximum air temperature and marine turtle stranding rates. In brief it shows most cases split responses were observed; when a split response was not noted stranding rates decreased and monthly mean maximum air temperature increased; there were similar response times noted for embayments and the whole blocks for monthly mean maximum air temperature; there were also similar response times for cumulative effect vs non-cumulative effect of monthly mean maximum air temperature.

QAIC's for all groups assessed were different and no patterns were observed (**Table 6.13**). In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.13. Model results for all turtles and monthly mean maximum temperature.

↑ denotes increased strandings rates with increased monthly mean maximum temperature. ↓ denotes decreased stranding rates with increased monthly mean maximum temperature. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	1-9↓	6	1-9↓	6	0-5↓,7-11↑	9	1-4↓,7-11↑	8,9
	SI	0-9↓,11-12↓	5,4,6	0-12↓	6,7,5,4	0-4↓,6-10↑,12↓	2,8,9	0-4↓,7-10↑,12↓	8,9
	LI	1-9↓	6	1-8↓	12,11	1-5↓,7-11↑	9,8	1-4↓,6-10↑	10,12,9
	A	2-10↓	6,7	1-10↓	6,7	1-5↓,7-11↑	9,3	1-4↓,6-10↑	9,8
	ALL IMM	0-9↓,11↓	5,6,4	0-12↓	6,7	0-4↓,6-10↑,12↓	8,3,9	0-4↓,6-10↑,12↓	8,9
	LARGE	1-10↓	6	1-9↓	6,7,11	1-5↓,7-11↑	3,9	1-4↓,6-10↑	9,8,10
-25	ALL	1-8↓,11↑	5,4	2-8↓,10-12↑	11,5,12,6	1-4↓,6-10↑,12↓	8	1-4↓,7-11↑	9,8
	SI	0-5↓,10↑	3,2,10	3-7↓,11↑	11,12,5,6	0-3↓,5-9↑,11-12↓	7	1-4↓,7-10↑	9,8,10
	LI	2-9↓	5,6,4	2-8↓,11-12↑	11,12	1-5↓,7-11↑	3,9	1-4↓,7-11↑	9,11,10
	A	2-9↓	6,5	0↑,3-8↓,11-12↑	11,12,5,6	1-5↓,7-11↑	9	2-5↓,7-11↑	9,8,3
	ALL IMM	0-7↓	4,3,5	2-8↓,11-12↑	11,12,5,6	0-4↓,6-10↑,12↓	7,8	1-4↓,7-11↑	9,8
	LARGE	2-9↓	6,5	2-8↓,11-12↑	11,12,5,6	1-5↓,7-11↑	3,9	1-5↓,7-11↑	9,3

Table 6.13 Continued

-23	ALL	1-12↓	5,4	0-12↓	12,4,3,5	1-4↓,7-9↑	2,3	1-3↓,6-8↑,12↓	2,12,1,3
	SI	0-8↓,11-12↓	3,2	0-12↓	12,4,3,5	0-3↓,6-9↑,12↓	2	0-3↓,6-8↑,12↓	2,1,12
	LI	-	-	-	-	-	-	-	-
	A	0↑,3↓,5-10↓	8,9	-	12,11,10	0↑,3-7↓,9-12↑	5	2-3↓	12,11,10
	ALL IMM	0-8↓,11-12↓	3,2	0-12↓	12,4,3	0-3↓,6-9↑,12↓	2	0-3↓,6-8↑,12↓	2,1,12,3
	LARGE	3-11↓	8,7,9	3-6↓		2-6↓,9-11↑	5,4	2-3↓	12,2,3,11
-19	ALL	0-12↓	12	0-12↓	12	0-3↓,7-8↑,11-12↓	12,1	0-3↓,6-7↑,11-12↓	12
	SI	0-12↓	12	0-12↓	12	0-3↓,6-8↑,11-12↓	1	0-3↓,6-7↑,11-12↓	1,12
	LI	0-12↓	12,11	0-6↓,9-12↓	12,11	0-3↓,12↓	12	0-2↓,12↓	12,1,2
	A	0-4↓,11-12↓	12	0-5↓,10-12↓	12	0-1↓,12↓	12	0-1↓,6↑,11-12↓	12
	ALL IMM	0-12↓	12	0-12↓	12	0-3↓,7-8↑,11-12↓	1	0-3↓,6-7↑,11-12↓	1,12
	LARGE	0-6↓,10-12↓	12	0-6↓,10-12↓	12	0-1↓,12↓	12	0-2↓,6↑,11-12↓	12

6.6.3.5. ***Monthly Mean Minimum Air Temperature***

Table 6.14 summarizes the relationship between monthly mean maximum air temperature and marine turtle stranding rates. In brief it shows for cumulative effects across all latitudes, as monthly mean minimum air temperature increased, stranding rates for all turtles decreased; non-cumulative effects across all latitudes split responses were noted; non-cumulative effects across all latitudes, there was an immediate decrease in strandings rates (0-5-month lag), followed by an increase (5-10-month lag) and then a decreased (11-12-month lag); within each examined latitudinal block, there were similar response times noted for embayments and the whole blocks for monthly mean minimum air temperature; were very obvious stranding differences between cumulative and non-cumulative effects. Non-cumulative effects resulted in split responses whereas in most cases the cumulative effects resulted in decreasing stranding rate with increasing mean minimum air temperature.

QAIC's for all groups assessed were different and no patterns were observed (**Table 6.14**). In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.14 Model results for all turtles and monthly mean minimum temperature.

↑ denotes increased strandings rates with increased monthly mean maximum temperature. ↓ denotes decreased stranding rates with increased monthly mean maximum temperature. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	1-9↓	6	1-9↓	6	1-4↓,7-11↑	9	1-4↓,7-11↑	8,9
	SI	0-8↓	4,5,6	0-9↓	6,5,7,4	0-4↓,6-10↑,11↓	8,9	0-4↓,6-10↑	8,9
	LI	1-9↓	6,7	1-9↓	12,11,10	1-4↓,7-11↑	9,9	0-4↓,7-10↑	10,12,9
	A	1-9↓	6	1-8↓	6,7	1-5↓,7-11↑	9	0-4↓,6-10↑	9,8
	ALL IMM	0-9↓	5,4,6	0-9↓	6,7	1-4↓,6-10↑,12↓	8,9	0-4↓,6-10↑	8,9
	LARGE	1-9↓,11↓	6	0-9↓	6,7	1-5↓,7-11↑	9	0-4↓,6-10↑	9,8,10
-25	ALL	0-8↓,12↓	4,5	1-10↓	5,6,4	0-4↓,6-10↑,12↓	2,8	1-4↓,7-10↑	9,2,8
	SI	0-6↓,12↓	12,2,3	1-10↓,12↓	5,6,4,7	0-3↓,5-8↑,11-12↓	12,7	1-4↓,7-10↑	9,2,8
	LI	1-8↓	5,4,6	0-9↓	12,7,11	1-4↓,6-10↑	8,9	1-4,7-10↑	9,10,8
	A	1-9↓	6,5,3	0-10↓	5,6,7,4	1-5↓,7-11↑	9	1-5↓,7-11↑	9,8,2
	ALL IMM	0-7↓,12↓	3,4,2	0-10↓,12↓	5,4,6,7	0-3↓,6-9↑,12↓	7,2,8	1-4↓,7-10↑	9,2,8
	LARGE	1-9↓	5,6	0-10↓	5,6,4,7	1-5↓,7-11↑	9	1-5↓,7-11↑	9,8

Table 6.14 Continued

-23	ALL	0-12↓	5,4	0-8↓,10-12↓	12,3,2,4	0-4↓,7-10↑,12↓	2	0-3↓,6-8↑,12↓	2,1,7
	SI	0-7↓,11-12↓	3,2	0-4↓,11-12↓	2,3,12	0-3↓,5-9↑,12↓	1	0-3↓,6-8↑,12↓	1,2,7
	LI	-	-	-	-	-	-	-	-
	A	0↑,3↓,5-10↓	8,9	3-6↓,10-12↓	12,11,10	0↑,2-6↓,9-12↑	10,5,11	2-3↓	12,11,10
	ALL IMM	0-8↓,11-12↓	3,2	0-7↓,10-12↓	12,3,2,4	0-3↓,6-9↑,12↓	1,2	0-3↓,6-8↑,12↓	2,1,7
	LARGE	3-11↓	8,7,9	1-12↓	12,11,10	2-6↓,8-11↑	10	1-3↓,8↑	2,12,3,8
-19	ALL	0-6↓,12↓	12	0-5↓,12↓	12	0-2↓,6-8↑,12↓	12	0-2↓,5-8↑,11-12↓	12,0
	SI	0-6↓,12↓	3,2	0-5↓,12↓	2,1,3	0-3↓,5-9↑,12↓	7,1	0-2↓,5-8↑,11-12↓	12,1,0
	LI	0-6↓,12↓	12	0-3↓,12↓	12,11,2,3	0-2↓,8↑	12,8,9	0-2↓,6-7↑,12↓	12,6,7
	A	0-2↓,11-12↓	2	0-3↓,12↓	12	0-1↓,11-12↓	12	0-1↑,5-7↑,11-12↓	12,0,11
	ALL IMM	0-6↓,12↓	12,3,4,2	0-5↓,12↓	2,12,1,3	0-3↓,6-9↑,12↓	1	0-2↓,5-8↑,11-12↓	0,12,1
	LARGE	0-3↓,11-12↓	12	0-3↓,12↓	12	0-1↓,12↓	12	0-1↓,5-7↑,11-12↓	12

6.6.3.6. *Monthly Average Daily Diurnal Air Temperature Difference*

Table 6.15 summarizes the relationship between monthly average daily diurnal air temperature difference and marine turtle stranding rates. In brief it shows that in most cases, a split response was observed for monthly average daily diurnal air temperature difference; adults and large turtles from the whole -19° block did not show significant stranding responses; within each examined latitudinal block, there were similar response times noted for embayments and the whole blocks.

QAIC's for all groups assessed were different and no patterns were observed (**Table 6.15**). In most cases, the QAICs corresponded with significant responses, with an exception for the age classes which did not produce a significant relationship.

Table 6.15. Model results for all turtles and monthly average daily diurnal temperature difference.

↑ denotes increased strandings rates with increased average daily diurnal temperature difference. ↓ denotes decreased stranding rates with increased average daily diurnal temperature difference. Ageclass abbreviations: ALL = all turtles, SI = small immature, LI = large immatures, A = adult-sized, ALL IMM= all immature sized animals (small + large), Large = all large turtles (large immatures + adult-sized). Time frame reported is month ranges where responses were noted. The values reported in qAIC are the months with the most significant qAIC value.

Latitude	Age class	Cum-Whole		Cum-Bay		Non Cum-Whole		Non Cum-Bay	
		Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC	Time Frame	QAIC
-27	ALL	1-8↑	6	1-8↑	6	1-4↑,7-11↓	9,8	1-4↑,7-11↓	8,9
	SI	0-6↑	12,11	1-6↑,10-11↓	6,7,5,4	0-3↑,6-10↓	8,9	1-4↑,6-11↓	8,9
	LI	1-8↑	12,6,11,7	0-8↑	12,11	1-4↑,7-11↓	9	0-3↑,8-9↓	9,12,11
	A	1-7↑,11-12↑	6,12,5	0-7↑,11−	6,7	1-4↑,7-11↓	9,8	1-3↑,6-10↓	8,9
	ALL IMM	0-7↑	4,5,3	0-7↑	6,7	0-4↑,6-11↓	8,9	0-4↑,6-11↓	9,8
	LARGE	1-8↑	6,4,5	0-8↑	6,7,11	1-4↑,7-11↓	9	0-3↑,6-10↓	9
-25	ALL	0-8↑,12	2,3	0-12↑	5,4,3,2	0-3↑,6-9↓,12↑	2,1	0-4↑,7-10↓,12↑	2
	SI	0-6↑,12↑	2,1,0	0-12↑	4,5,3	0-2↑,5-8↓,11-12↑	12	0-4↓,8-9↓,12↑	12,0,1
	LI	0-7↑	2,3,4	1-9↑,12↑	12,11,4	0-3↑,6-10↓,12↑	8,7,9	0-4↓,6-10↓,12↑	12,9,8
	A	0-8↑	4,5,3	0-12↑	5,4,6,3	0-4↑,7-10↓,12↑	9,8,2	0-5,7-9↓,12↑	2
	ALL IMM	0-7↑	2	0-12↑	4,3,5,2	0-3↑,5-9↓,12↑	12,7,1	0-4↑,7-10↓,12↑	12,2,9,1
	LARGE	0-8↑	3,4,2	0-12↑	5,4,3,2	0-4↑,6-10↓,12↑	8,9	0-4↑,7-10↓,12↑	2

Table 6.15 Continued

-23	ALL	0-4↑	12,11,10	0-2↑	2,12	0-1↑,5-9↓	8,9,5,6	0-1↑,5↓	12,9,5,7
	SI	0-3↑	12,11,10	0-2↑,9-12↓	12,11,10	0↑,5↓	5	0↑,5↓,7↓,9↓12↓	5,9,12
	LI	-	-	-	-	-	-	-	-
	A	11-12↓	12	2-6↑	11,12,10	3-4↑,6-11↓	10,9,8,12	2↑	2,12,11
	ALL IMM	0-3↑	2,12,1	0-2↑,10-12↓	12,11,10	0-1↑,5-6↓	5	0↑ 5-7↓,9↓,12↓	9,7,12
	LARGE	2-4↑	12	1-5↑	12,11,3	1↑,6-10↓	8,10,9,12	2↑	2,10,12,11,9
-19	ALL	0-3↑,8-12↓	10,11	0-2↑,7-12↓	10,9	0-2↑,5-9↓	8,7	0-1↑,4-8↓	7,6
	SI	0-3↑,7-12↓	10,11	0-3↑,7-12↓	10,11,9	0-2↑,5-9↓	7	0-1↑,4-9↓	7,6
	LI	10-11↓	11,12,10	8-11↓	10,9,11,8	6↓,8↓	8,9,6	5-6↓	6,8,5,7
	A	-	12,9,11,10	0↑,7-10↓	8,9,10,11	-	12,11	0↑,5-6↓	5,11,7
	ALL IMM	0-3↑,7-12↓	10,11	0-3↑,7-12↓	10,9,11	0-3↓,5-9↓	8,7	0-1↑,4-9↓,12↑	6,7
	LARGE	-	10,11,9	0↑,7-11↓	9,10,8	-	12,8	0↑,4-7↓	5,6,7

6.7. Discussion

This is the first study of its kind to elucidate the effects of individual environmental variables on the stranding rates of coastal marine turtle populations and provides a baseline for future predictive models that can be used as real-time management tools. We found that strandings occurred after a lag phase, with water discharge having the greatest effect on stranding numbers. This study found that the cumulative effects of freshwater discharge in all latitudes resulted in increased strandings 7-12 months later (**Table 6.3 - 6.5 and Table 6.10 -6.12)**. The cumulative effects of mean maximum and minimum air temperature resulted in decreased stranding rates immediately through to a lag of 9 months (**Table 6.6 - 6.7, Table 6.13 - 6.14**). Monthly average daily diurnal air temperature difference resulted in increased strandings immediately through to a lag of 8 months (**Table 6.8 - 6.15**). There was an overall decrease in stranding rates 2-8 months after high rainfall events, although the relationship was less clear (**Table 6.2 - 6.9**).

When comparing cumulative effects against non-cumulative effects, non-cumulative effects were more likely to produce split or dual responses **(Table 6.2 - 6.15).** This could be due to cumulative effects having a more lasting, stronger effect. The cumulative effect of multiple months of increased discharge and rainfall potentially does not allow time for the seagrasses to recover and hence have a stronger effect on marine turtles through their diets.

When analysing latitude along the Queensland coastline, there was no evidence that stranding rates were different in different latitude, although there were some noticeable differences **(Table 6.2 - 6.15).** When comparing the effect of latitude for discharge, the -25° block produced split responses whereas the other blocks produced single responses (**Table 6.3** - **Table 6.5**, **Table 6.10** - **6.12)**. Although stranding rates at different latitudes responded differently the overall pattern of lagged stranding was similar, suggesting the increase in marine turtle stranding was not just a local issue rather, at least, a state-wide issue that occurred and warranted consideration at a state or larger regional level. Given the migratory pattern of marine turtles and their ability to move to new sites before returning to their within the broader range of their individual home sites (Shimada et al., 2016a), mitigation needs to consider widespread impacts and not just local habitats of known marine turtle populations.

When analysing age classes across the variables, there were no observed patterns in relation to which group responded first for each variable **(Table 6.2 - 6.15)**. This was not expected, it was expected that small immatures would be more susceptible to changes in dietary availability and would show responses before other age classes.

Embayments, when compared to the whole latitudinal block, did not influence the pattern of strandings but did decrease the lag phase for each examined environmental variable **(Table 6.2 - 6.15).** This could indicate that embayments are areas of concentrated discharge which is not dissipating in to the wider area, thus having an increased negative effect on the turtles and the aquatic vegetation for which they depend.

An interesting outcome from this study was that, while the response trends were the similar, green turtles as a group tended to respond ~ 1 month earlier than all turtles **(Table 6.2 - 6.15).** The reasons for this earlier response are uncertain, but may be related to diet. However, the small sample sizes of the other species prevented this trend being statistically analysed further.

As with any exploratory modelling, we identified several limitations that may influence the accuracy of any developed model including distributed sample equality, equal adequate sample sizes for each species, availability of environmental data such as seagrass abundance, habitat type and the distance offshore that an event was recorded. One of the limitations of these models is that the stranding sample size was different for each examined latitudinal block. Larger sample size may make the relationships more noticeable than smaller sample sizes, but as this used one of the longest running and largest datasets available, this may be difficult to correct. The -27° block recorded the most number of strandings over the study period (**Table 6.1**). This latitudinal block encompasses Moreton Bay which is known to support large fields of seagrass, other aquatic vegetation and a significant human population. The -23° block recorded the least number of strandings over the study period for a recognized hotspot (**Table 6.1**).

The model may have been strengthened by the use of food availability/viability as a factor. However, due to the paucity of data, it was decided to use weather as a proxy to this as weather data is available in immediate time. There is evidence that discharge and rainfall are adequate proxies for seagrass abundance as large-scale seagrass die-off have been closely associated in time and intensity to flooding (Poiner et al., 1993b; Preen et al., 1995; Wetz and Yoskowitz, 2013). This study may also have been strengthened by determining if different species showed different responses times and directions. This was not possible due to the small sample sizes of the other species occurring within the study location.

Within coastal waters green turtles are almost exclusively herbivorous, feeding principally on seagrass and a wide range of algae and mangrove fruits (Limpus, 2008a; Read and Limpus, 2002). Occasionally, green turtles feed on macroplankton, including jellyfish, bluebottles, small crustaceans and dead fish (Limpus, 2008a; Read and Limpus, 2002). Brand-Gardner et al.(1999) found that within Moreton Bay small immature green turtles forage selectively on plants with higher nitrogen levels and lower levels of fiber (such as *Gracilaria* sp.). Due to this strong dependency on aquatic vegetation, it has meant green turtles that live within inshore coast habitats where aquatic vegetation is a large component of their diet have suffered during and post the extreme weather events, such as the flooding in Queensland in 2010-11.

This study has identified that there are relationships between specific environmental variables (freshwater discharge and air temperature) and marine turtle strandings. These findings will allow first responders to be more prepared for increases in strandings following increases in freshwater discharge rates. These models can be used to form the basis for an exploratory model which can be used to predict future responses to adverse weather events including increased freshwater discharge, increased rainfall and changes in mean air temperature.

This article is a desktop analysis and does not contain any studies with animals.

We would like to thank the Queensland Marine Wildlife Strandings and Mortality network, EHP and all contributors to the StrandNet database. The authors would like to thank Dr.

Julien Martin for his assistance and guidance with the modelling aspects of this study. The authors would also like to thank Dr. Milani Chaloupka who also provided confirmation on the modelling aspects.

Chapter 7. Predicting the magnitude of marine turtle strandings based on weather conditions

This chapter provides predictive modelling based on the relationships determined in the previous chapter. This chapter will provide first responders with resources to help them be better prepared for increased in marine turtle strandings after extreme weather events, by providing them with magnitudes of strandings to expect and at what time frame to expect these increases.

This chapter forms the basis of a predictive model that will be further refined and developed for use by coastal resource managers.

7.1. Abstract

During recent decades, on average, between 500 and 800 marine turtles have stranded annually along the Queensland coastline (Biddle and Limpus, 2011). In 2011, following a large cyclone and protracted state-wide flooding, there were over 1793 stranded marine turtles reported in Queensland in the Queensland Environment and Heritage Protection StrandNet database. Meager and Limpus (2012) stated that the most plausible explanation for the high rate of strandings and mortalities of near shore green turtles during 2011 was extreme weather events that occurred in late 2010 and early 2011, which impacted seagrass and other foraging areas.

We developed a predictive model to enable first responders to anticipate in advance when increases in stranding numbers are likely to occur under a range of environmental conditions including air temperature and water discharge. This model is a prototype that has been tested only one latitudinal area and has only been tested on the total number of strandings.

When looking at the significant values of the relationships it is apparent that some variables are more closely aligned to stranding numbers than others. Several different models were trialled, however the cumulative mean cumec discharge of freshwater from waterways provided the most accurate correlation with stranding rate. We also demonstrated that discharge and air temperature may be among the key factors affecting stranding rates in Queensland; although the current model may be improved by including more variables such as aquatic vegetation availability and viability or cyclonic habitat damage.

The Queensland and Commonwealth Governments have identified that the runoff from rivers is one of the most significant impacts on inshore coastal habitats (coral reefs and seagrass). Their response to this identification was the “Reef 2050 Long-Term Sustainability Plan” which aims to reduce sediment and chemical outflow. This rationale of this major management initiative correlates with the findings of this study.

This predictive model will allow first responders and marine resource managers to be better prepared for increases in marine turtle stranding numbers. It will enable them to

conduct in-depth health investigations into cause of death/stranding, get more animals to triage/rehabilitation and develop a thorough understanding of disease processes to hopefully reduce these pressures in the future.

7.2. Introduction

During recent decades, on average, between 500 and 800 marine turtles have stranded annually along the Queensland coastline (Biddle and Limpus, 2011). In 2011, following a large cyclone and protracted state-wide flooding, there were over 1793 stranded marine turtles reported in Queensland in the Queensland Environment and Heritage Protection StrandNet database. This was the largest annual number of turtles reported stranded in the 16 years for which comprehensive data has been collected for this region (Meager and Limpus, 2012b). Rainfall and freshwater discharge as a consequence of this flooding and cyclonic coastal habitat damage are closely linked with food availability that became severely depleted in the months following, as well as air temperature may have all played a role in this mass mortality.

This increase in the number of stranded turtles raised much public interest and action over responding to turtles challenged by adverse weather events in an attempt to minimise the negative effect of natural disasters and maximise the number of turtles that survive these catastrophic periods.

Meager and Limpus (2012) stated that the most plausible explanation for the high rate of strandings and mortalities of near shore green turtles during 2011 was extreme weather events that occurred in late 2010 and early 2011, which impacted seagrass and other foraging areas. They linked this because most of the examined mortalities (92% of identifiable natural causes of death in turtles) were attributed to protracted ill health/poor body condition in green turtles and dugongs; which both primarily forage on aquatic vegetation. There was evidence that seagrass pastures, coral reefs, mangrove forests and algal beds in Queensland were impacted by elevated rainfall, flooding and a cyclone during the summer of 2010/2011 (Great Barrier Reef Marine Park Authority, 2011b). Meager and Limpus (2012) also stated that elevated rates of turtle and dugong mortalities have occurred following similar weather events in the past, for example in 1992 when 99 dugongs stranded in Hervey Bay (Preen and Marsh, 1995).

Another factor that may affect our understanding of marine turtle survival is the environmental temperature. Within Moreton Bay, loggerhead and green turtles are captured during winter on the intertidal banks where the water temperature has been recorded as low as 15 °C in winter (Limpus and Limpus, 2003; Read et al., 1996) and as high as 27 °C during summer. Through the ongoing monitoring studies of recaptured animals and satellite telemetry there is no evidence that the east Australian loggerhead or green turtle populations undertake north-south, summer-winter nonbreeding migrations (Limpus and Limpus, 2003; Read et al., 1996) and therefore these populations likely endure these extreme temperatures. This is in comparison to other populations of marine turtles in the northern hemisphere which exhibit migration during cooler months (Carr and Caldwell, 1956; Musick et al., 1997; Witherington et al., 2006).

Marine turtle stranding numbers follow seasonal trends influenced by weather events as well as land-based and at-sea seasonal activities, with links made between extreme weather and increased strandings (Flint et al., 2015; Marsh and Kwan, 2008; Meager and Limpus, 2012b; Preen and Marsh, 1995). More specifically, Meager and Limpus (2014) proposed links between periods of elevated freshwater discharge, low air temperatures and increased dugong mortality. They found that 9 months after elevated freshwater discharge there was an increase in dugong mortality.

The monitoring of marine vertebrates including turtles at sea can be expensive. The use of strandings can be an effective ancillary tool to provide minimum counts of at sea mortality and threats (Peltier et al., 2012). In turn, the creation and use of predictive models has proven helpful in other species in allowing first responders to have a better understanding of causes of strandings, and be better prepared for future stranding events (Meager and Limpus, 2014). When analysing marine turtle stranding numbers there is high variability between years. Due to this variability and the proposed link between marine turtle strandings and environmental variable it would be advantageous to quantify these known influences of stranding rates so first responders have an idea of how high stranding numbers are likely to be and when it is likely to occur after an abnormal environmental event.

We developed a predictive model to enable first responders to anticipate in advance when increases in stranding numbers are likely to occur under a range of environmental (temperature, water discharge and rainfall) conditions. By combining previously published individual models Flint et al., (2017b)(Chapter 6), we developed a single model to allow users to input current environmental variables and determine the resultant stranding peak (lag and intensity).

7.3. Methods

7.3.1. Data

7.3.1.1. *Stranding Data*

StrandNet is the Queensland Government's Department of Environment and Heritage Protection (EHP) state-wide database which records dead, sick and injured threatened marine animals for the entire coast of Queensland and adjacent Commonwealth waters. Records are received from members of the public, and employees of EHP, Queensland Parks and Wildlife (QPWS), Queensland Department of Agriculture, and Fisheries (DAF) and the Great Barrier Reef Marine Park Authority (GBRMPA). Information is collated and stored in this central database. Once reports are entered by on-ground staff the information available is verified by regional and state coordinators for standardization.

7.3.1.2. *Location*

The study area encompassed latitude -10.78° to -28.16° and longitude 142.15° to 155°. The east coast of Queensland was selected as it has a long-term and complete dataset; with data collection biased to regions of survey and higher populations. This limitation is openly acknowledged by Meager and Limpus (2012) but considered valid as a representative of minimum recovery rate and indicative of trends occurring. As the exact location where a stranding was reported was not necessarily where the impact/incident occurred, strandings were grouped into latitudinal blocks of 1° to more accurately address this potential error. The data used to produce these models was limited to the hotspot recognized by Flint et al. (2015) as consistently having the largest number of strandings along the Queensland coastline, 27° latitude. As responses were similar between the embayment and the whole latitudinal block, the whole block was chosen as it represented the largest number of reported strandings.

7.3.2. Environmental Data

Freshwater discharge is the amount of freshwater running through the river recording stations, measured in cumecs (cubic meter per second, m^3s^{-1}). Freshwater discharge data was downloaded from the Department of Natural Resources and Mines (https://water-monitoring.information.qld.gov.au/) under the Creative Commons Attribution 3.0 Australia (CC BY) license. Discharge data from the most downstream gauging station for each major drainage area within the latitudinal block was selected. Data for each month between 1996 and 2013 was analysed.

Temperature and rainfall data was obtained from the Bureau of Meteorology (http://www.bom.gov.au/climate/data/). Rainfall and temperature data was obtained from a central station within each latitudinal block with a complete dataset. Mean monthly maximum and minimum temperatures were used. The average diurnal temperature change was calculated by obtaining the maximum and minimum daily temperatures and determining the difference then averaging over the month. Data for each month between 1996 and 2013 was analysed.

7.3.3. Modelling

The initial modelling was performed as outlined in Flint et al., (2017b). In brief, models were run with all variables (cumulative mean discharge, mean discharge, peak discharge, rainfall, average daily diurnal temperature difference, monthly mean maximum temperature, monthly mean minimum temperature and rainfall) combined. These models proved non-significant ($p>0.1$); therefore, each environmental factor was run separately to determine the individual effect.

Each environmental factor was modelled separately to determine its individual effect. In order to compare models for best fit, QAIC weight were calculated using the relative likelihood of the model. QAIC is the quasi Akaike Information Criterion (AIC). QAIC weights allow for the selection of a “best approximating model” (Burnham and Anderson, 2002). This was then used in conjunction with the significance of the variables to determine the model which best explained the most variance in the data. Significance was set at <0.1.

These independent models were run separately and combined to create numerous predictive model in this study to determine the "best one".

7.3.4. Predictive model

The predicative model was created using the best fit model identified for each environmental variable with the QAIC value used to determine the best lag period. Quasipoisson distribution was used due to over dispersion.

The below equations were chosen based on the above criteria as the best fit models:

Equation 1: $glm(TOTAL\sim C\_M\_Clag\_8 + M\_Maxlag\_6 + M\_Minlag\_6 + AV\_D\_CHlag\_6, na.action = na.omit, family = quasipoisson)$

Equation 2: $glm(TOTAL\sim C\_M\_Clag\_8 + M\_Maxlag\_6, na.action = na.omit, family = quasipoisson)$

Equation 3: $glm(TOTAL\sim C\_M\_Clag\_8 + M\_Minlag\_6, na.action = na.omit, family = quasipoisson)$

Equation 4: $glm(TOTAL\sim C\_M\_Clag\_8 + AV\_D\_CHlag\_6, na.action = na.omit, family = quasipoisson)$

Equation 5: $glm(TOTAL\sim C\_M\_Clag\_8, na.action = na.omit, family = quasipoisson)$

Equation 6: $glm(TOTAL\sim M\_Maxlag\_6, na.action = na.omit, family = quasipoisson)$

Equation 7: $glm(TOTAL\sim M\_Minlag\_6, na.action = na.omit, family = quasipoisson)$

Equation 8: $glm(TOTAL\sim AV\_D\_CHlag\_6, na.action = na.omit, family = quasipoisson)$

C_M_Clag_8 represents the cumulative mean discharge with an 8-month lag, M_Maxlag_6 represents the monthly mean maximum temperature with a 6-month lag, M_Minlag_6 represents the monthly mean minimum temperature with a 6-month lag, AV_D_CHlag_6 represents the average diurnal daily difference with a 6-month lag. Total is the total number of marine turtles reported stranded each month.

In order to create a predictive model, the following general equation was used:

$$exp(glm\$coefficients[1] + tglm\$coefficients[2] * variable)$$

Using this equation different values for the variable: C_M_C lag_8, M_Maxlag_6, M_Minlag_6 and AV_D_CH_6 were used to test the robustness and feasibility of the resultant outputs. For example, the predictive model for equation 1 would be:

$$exp(glm\$coefficients[1] + glm\$coefficients[2] * ((C\_M\_Clag\_8 + M\_Maxlag\_6 + M\_Minlag\_6 + AV\_D\_CHlag\_6))$$

As a trial, all environmental variables were tested as minimum values, as maximum values and average values.

7.3.5. Weighted predictive model

As a further investigation, the variables were weighted to determine if the variables had different levels of effect on the stranding rate. Different weighting values were trialed based on the authors' experience and understanding of marine turtle ecology under varying influences and the models tested previously in Flint et al., (Submitted).

Equation 1 was tested three times with different weights for each variable (Table 7.1).

Table 7.1 Weights used to test weighted model for equation 1.

	Weight 1	**Weight 2**	**Weight 3**
cumulative mean discharge with 8-month lag	0.8	0.7	0.9
monthly mean maximum temperature with 6-month lag	0.05	0.1	0.005
monthly mean minimum temperature with 6-month lag	0.05	0.1	0.005
average diurnal daily difference with 6-month lag	0.1	0.1	0.09

Equations 2- 4 were tested three times with different weights for each variable as set out in **Table 7.2**.

Table 7.2. Weights used to test weighted model for equation 2-4. Each of the variables M_Maxlag_6, M_Minlag_6 and AV_D_CHlag_6 were tested individually.

	Weight 1	**Weight 2**	**Weight 3**
cumulative mean discharge with 8-month lag	0.8	0.9	0.95
monthly mean maximum temperature with 6-month lag			
monthly mean minimum temperature with 6-month lag	0.2	0.1	0.05
average diurnal daily difference with 6-month lag			

7.4. Results

To test the models, the average, minimum and maximum number of strandings to occur in -27° latitudinal block were calculated using the stranding database, results are set out in **Table 7.3** .

Table 7.3. The maximum, minimum and average number of strandings recorded in the -27° block.

	Number of strandings recorded
Minimum	3
Maximum	107
Average	24

Testing the models produced different results for each model, with varying levels of significance (**Table 7.4**). All variables in equation 1 produced significant results ($p>0.1$). For equations 2-5 cumulative mean discharge with a lag of 8 was the only significant variable ($p>0.1$). For equations 6-8there were no significant variables noted.

Table 7.4. p values for each model tested.

Equation	Variable	p value
1	cumulative mean discharge with 8-month lag	0.000343
	monthly mean maximum temperature with 6-month lag	0.093714
	monthly mean minimum temperature with 6-month lag	0.097599
	average diurnal daily difference with a 6-month lag	0.095978
2	cumulative mean discharge with 8-month lag	0.000148
	monthly mean maximum temperature with 6-month lag	0.235297
3	cumulative mean discharge with 8-month lag	0.000199
	monthly mean minimum temperature with 6-month lag	0.242436
4	cumulative mean discharge with 8-month lag	0.000253
	average diurnal daily difference with a 6-month lag	0.356158
5	cumulative mean discharge with 8-month lag	0.00012
6	monthly mean maximum temperature with 6-month lag	0.234
7	monthly mean minimum temperature with 6-month lag	0.193
8	average diurnal daily difference with a 6-month lag	0.217

7.4.1. Predictive model

Without weighting any of the variables, the prediction using this model with all environmental variables set as averages (Equation 1) (normal expected conditions) resulted in 21 strandings within the block per month. Simulating this model with all environmental variables set as minimums resulted in 20 strandings within the block per month (**Table 7.5**). Simulating this model with all environmental variables set as maximum resulted in 57 strandings within the block per month.

7.4.2. Weighted predictive model

In **Table 7.5** each environmental variable was set as the minimum value recorded in the discharge data (min), maximum value recorded in the discharge data (max) and the average value recorded in the discharge data (ave). Using the different weights in each equation produced different numbers of predicted strandings (**Table 7.5**).

Table 7.5. Results of predictive models, with different weights and variables used.

	Environmental variables set as	C_M_Clag_8 Weight	M_Maxlag_6 Weight	M_Minlag_6 Weight	AV_D_CHlag_6 Weight	Predicted strandings
Equation 1	Min	-	-	-	-	21
	Max	-	-	-	-	57
	Ave	-	-	-	-	21
Equation 1	Min	0.7	0.1	0.1	0.1	20.46
	Max	0.7	0.1	0.1	0.1	40.96
	Ave	0.7	0.1	0.1	0.1	20.88
	Min	0.8	0.05	0.05	0.1	20.455
	Max	0.8	0.05	0.05	0.1	45.137
	Ave	0.8	0.05	0.05	0.1	20.91
	Min	0.9	0.005	0.005	0.09	20.44
	Max	0.9	0.005	0.005	0.09	49.78
	Ave	0.9	0.005	0.005	0.09	20.95
Equation 2	Min	0.8	0.2	-	-	16.71
	Max	0.8	0.2	-	-	38.4
	Ave	0.8	0.2	-	-	17.1
	Min	0.9	0.1	-	-	16.7
	Max	0.9	0.1	-	-	42.53
	Ave	0.9	0.1	-	-	17.14
	Min	0.95	0.05	-	-	16.69
	Max	0.95	0.05	-	-	44.73
	Ave	0.95	0.05	-	-	17.15

Table 7.5 Continued

Equation 3	Min	0.8	-	0.2	-	20.66
	Max	0.8	-	0.2	-	46.95
	Ave	0.8	-	0.2	-	21.12
	Min	0.9	-	0.1	-	20.65
	Max	0.9	-	0.1	-	51.95
	Ave	0.9	-	0.1	-	21.19
	Min	0.95	-	0.05	-	20.65
	Max	0.95	-	0.05	-	54.65
	Ave	0.95	-	0.05	-	21.22

Equation 4	Min	0.8	-	-	0.2	28.4
	Max	0.8	-	-	0.2	64.3
	Ave	0.8	-	-	0.2	29.13
	Min	0.9	-	-	0.1	28.4
	Max	0.9	-	-	0.1	71.22
	Ave	0.9	-	-	0.1	29.02
	Min	0.95	-	-	0.05	28.45
	Max	0.95	-	-	0.05	74.91
	Ave	0.95	-	-	0.05	29.237

For equations 5-8, each environmental variable was set as the minimum value recorded in the discharge data (min), maximum value recorded in the discharge data (max) and the average value recorded in the discharge data (ave). This produced different predicted stranding rates compared with those presented in **Table 7.6**.

Table 7.6 Results of predictive models, with different variables used.

		Environmental variables set as	Predicted strandings
Equation 5	cumulative mean discharge with 8-month lag	Min	23.9
		Max	68.97
		Ave	24.64
Equation 6	monthly mean maximum temperature with 6-month lag	Min	22.81
		Max	26.7
		Ave	24.72
Equation 7	monthly mean minimum temperature with 6-month lag	Min	22.29
		Max	26.66
		Ave	24.79
Equation 8	average diurnal daily difference with a 6-month lag	Min	26.82
		Max	21.44
		Ave	24.71

7.5. Discussion

This study has developed the first predictive models for assessing the impact of adverse weather conditions and catastrophic events on marine turtle strandings, using the largest long term dataset currently available. This study used Moreton Bay as a beta-test site due to the larger sample size of stranded marine turtles recorded in this latitude (Flint et al., 2017b), significant land use changes, increasing development related to population growth, supporting high human population which is increasing and decline in ecosystem health (Gibbes et al., 2014) and being a significant foraging ground for the southern Great Barrier Reef green turtle population (Shimada, 2015).

These model uses empirical data to support the hypothesis by Flint et al., (2017b; 2015); Marsh and Kwan, (2008); Meager and Limpus, (2012b) to demonstrate there is a predictable link between weather events and stranding rates.

When looking at the significant values it is apparent that some variables are more closely aligned to stranding numbers than others (**Table 7.4**). This was expected as we used QAIC weights to determine the best fit model.

Testing the unweighted model using all variables combined and comparing the predicated stranding rates (**Table 7.5**) to the actual stranding rates (**Table 7.3**), the minimum values were overestimating the strandings occurring, the average values were underestimating the strandings occurring and the maximum values were grossly underestimating the strandings that were occurring (~50%).

As such, we tried developing the weighted model using all variables combined and comparing the output to the predicted stranding rates. In all cases, the predicted stranding rates and the actual stranding rates were not closely aligned. When looking at the weighted model with equations 2 and 3, again the predicted stranding rates and the actual stranding rates were not aligned. Assessing each variable (equations 6-8), produced values that were noticeable different to the actual stranding rates.

The weighted and unweighted equation 1-3,6-8 were ruled out as acceptable predicative models. Possible reasons for this is that not all variables are interacting evenly to influence stranding rates or that not all variables are interacting together.

From the cumulative mean cumec discharge model (equation 5), the predicted marine turtle stranding rate was underestimated for the average values; was close to predicting the average stranding rate and underestimated the maximum values.

Each of these models demonstrated the predictive accuracy of the resultant output is not simply a linear relationship to one particular input or combination of all the inputs, rather a selective weighted combination of some variables. We demonstrated discharge and air temperature may be among the key factors affecting stranding rates in Queensland.

Running the weighted model with equation 4, the minimum values and the average values for strandings were overestimates and the maximum value was underestimated. However, the estimates were closer to the actual values than the other models. This indicates that this model produces the closest values of all model iterations we tested. We further refined this model by weighting cumulative mean cumec discharge with a weighting of 0.95 and the average daily diurnal temperature difference as a weighting 0.05. This model predicts (i) the maximum number of strandings which might occur under the minimum recorded environmental variables; (ii) the minimum number of strandings which might occur under the maximum environmental conditions; and (iii) the approximate number of strandings which occur under the average environmental conditions. This provides a significant advantage to those charged with responding to strandings- first responders are able to not only know the number of strandings to expect under average conditions, but this model also produced the minimum number of strandings to expect under maximum adverse environmental conditions and the maximum number of strandings to expect under minimum adverse environmental conditions. This allows management teams to budget resources for disasters well in advance of the actual need.

We only examined three basic input factors for this model. It may be strengthened by the incorporation of additional variables such as aquatic vegetation availability and viability or the degree of cyclonic habitat damage. Due to the immediate paucity of data, it was not possible to include data for these variables, as often sea grass density is determined as a proxy of, or demonstrated as a direct link to, known weather (creating a cyclical variable if included). For example, large-scale seagrass die-off have been directly attributed to flooding in the past (Poiner et al., 1993b; Preen et al., 1995) which would artificially increase the weighting of the freshwater discharge and rainfall variables already being modelled. Similarly, cyclonic habitat damage is not quantified for many of the areas we studied and could not be incorporated into the model as a value. These types of limitations warrant further investigation to incorporate such important variables.

These models were also limited by only analysing the highest value QAIC value and not using the full number of lag periods. This method was selected as it uses the lag periods where the most variation is mathematically explained by the model, but this approach has the potential to create problems in that although the QAIC value was the highest, there was not always a strong significant relationship between these variables.

Marine turtles are seen as flagship species for conservation (Tisdell and Wilson, 2003) and thus their conservation is important for a variety reasons particularly to protect ecological, aesthetic, economic, existence and bequest values (Aguirre and Lutz, 2004; Chaloupka et al., 2008b; Feck and Hamann, 2013; Jackson et al., 2001; Tisdell and Wilson, 2001). Being able to predict the response of stranding numbers to a range of identified environmental variables will contribute to the recovery of critical habitat as well as assist with the conservation of the species through increased and improved awareness.

The Queensland and Commonwealth Governments have identified that the runoff from rivers is one of the most significant impacts on inshore coastal habitats (coral reefs and seagrass)(Great Barrier Reef Marine Park Authority, 2015, 2014). They have responded with the "Reef 2050 Long-Term Sustainability Plan" with the aspiration of reducing sediment and chemical outflow from our rivers to encourage a shift of management from regional to catchment zones for improving the quality of inshore coastal habitat (Great Barrier Reef Marine Park Authority, 2015). The rationale underpinning this major management initiative correlated with the findings of this study that demonstrated discharge to be the biggest influence on stranding rate of marine turtles, albeit lagged by several months. This relationship is believed to be so potent due to river outflow having the ability to alter nearshore habitat (e.g. remove or kill of sea grasses and increase sedimentation) in a relatively short period of time.

If first responders are better prepared to respond to strandings, in turn they can treat more animals with the aim of returning them back to the wild in a timely manner. It will also enable them to conduct in depth health investigations into cause of death and stranding and underlying health issues within the population. This will develop a more thorough understanding of disease processes with the hope of reducing future increases in stranding and finding any potential point sources and causes of the disease processes.

This model has enabled us to identify gaps in our current datasets and understanding of the interplay between these environmental variables.

Despite the limitations to this model and the scope for further refinement, the weather conditions currently being experienced are unprecedented in the 41 year history of the Great Barrier Reef Marine Park and much of Queensland as a whole (Great Barrier Reef

Marine Park Authority, 2011a; Steffen et al., 2013) and warrant some type of predictive management tool to assist mitigation strategies. As such this model will play an important role in management of such events and will evolve in predictive accuracy as other variables such as weighted food availability, potential habitat damage and multiple lag phases can be built into the presented framework.

Chapter 8. Discussion and Conclusion

8.1. Introduction

Rehabilitation of stranded marine turtles along the coastline of Queensland, Australia, has become an increasingly popular approach to help with the recovery of local marine turtle populations (Feck and Hamann, 2013). Following catastrophic environmental flooding events in 2011 and 2012 that resulted in record numbers of marine turtle and dugong strandings, many new rehabilitation facilities opened. Support for these ventures were a mixture of government and private donation funded operations (Feck and Hamann, 2013), which tended to provide autonomy but the rush to address the significant influx of injured/sick/debilitated animals resulted in a variety of standards and operating procedures. As such, the effectiveness and benefits of rehabilitation was highly variable as each facility had to manage high operating costs to attain moderate and often vague outcomes.

Through the assessment of stranding records, rehabilitation data and recapture results, we proposed that rehabilitation may not be the most effective tool in the conservation of marine turtles in Queensland, Australia. However, rehabilitation still offers several advantages to warrant its continued role in marine turtle health and conservation. It contributes to the health and survival of individual animals, which is an important contribution to any endangered species. It provides a resource to biologists and veterinarians struggling to understand the key diseases impacting local marine turtle populations. It also functions as an invaluable vehicle to increase conservation awareness and conduct outreach to the community to help the public understand ways they can help save marine turtles.

In an attempt to assess available options, this study quantified the effectiveness of rehabilitation of marine turtles in Queensland, in comparison to alternative strategies, such as release without treatment or the use of temporary mobile hospitals responding to specific stranding events. Further, the available data was modelled to determine trends of “who, what, when and where” strandings may occur. Using this platform, a predictive model was created to determine what environmental trigger points are required to initiate strandings and what resources would be needed given the magnitude of the environmental catastrophe.

8.2. General Hypothesis

This investigation began to address the limited knowledge available about the success of marine turtle rehabilitation after extreme weather events by investigating the hypothesis that *rehabilitation is a viable practice used to successfully treat and return green and loggerhead marine turtles to their resident grounds after a catastrophic event* through a series of objectives.

The general hypothesis was only partially upheld despite each objective of this investigation being met. The small sample size of animals which were recaptured after release from rehabilitation prevented a complete an analysis of the success of rehabilitation post extreme weather event.

Trends in marine turtle stranding rates were identified (Chapter 3, Flint et al., 2015), which led to further investigation into the patterns shown in marine turtle stranding rates (Chapter 6, Flint el al., submitted). From these patterns, a predictive model was developed to enable management agencies to be better prepared to deal with increased stranding numbers by knowing when the increases are likely to occur and how high the numbers are likely to be (Chapter 7).

In conjunction with these analyses, parallel studies were conducted into the success of marine turtles that were sent to rehabilitation (Chapter 4, Flint et al., 2017), compared to those animals which were not sent to rehabilitation (Chapter 5, Flint et al., 2017b).

8.3. Objectives

8.3.1. Objective 1: Examine the current literature to understand current state of knowledge for marine turtle stranding trends and the links between extreme weather events (Chapter 1).

No gap analysis had been performed assessing the trends on marine turtle strandings through the years and how this may relate to extreme weather events. Further, no analysis had been completed into the success of rehabilitation of marine turtles in Queensland. These two gaps were vital to fill if we were to continue practicing marine turtle rescue and rehabilitation to share knowledge, assist in creating a uniform approach to these practices and use this information to make informed management decisions on environmentally induced strandings. This study used available data through StrandNet to determine the patterns of stranding in Queensland based on species, age class, time of year, location and cause to enable resources to be better assigned and that species conservation objectives were being met.

8.3.2. Objective 2: Examine the strandings between 1996 and 2013 to get a better understanding of the trends and cycles (Chapter 3).

We found definite patterns in the stranding of marine turtles along the coast of Queensland. Green turtles were also the most frequent species and immature animals were the most frequent age class sent to rehabilitation both by numbers and by proportion. Analysis showed that turtle stranding is cyclical across years with more turtles stranding during the months coming out of winter (August to November) and fewer turtles stranding in the months when waters start to cool (April to June).

Four location hotspots near Brisbane, Gladstone, Rockhampton and Cairns were identified that received the majority of reported stranding after extreme weather events and the reasons why as well as options to mitigate became the focus of this investigation. Not surprisingly, these landmarks are the ocean outflow points of the four major catchment basins in Queensland. When high rainfall events cause flooding, collected freshwater flows from inland to discharge at these sites. Brisbane, Gladstone and Cairns have been established as large rehabilitation centre hubs, with other smaller facilities located in surrounding areas.

Depending on the specific causes and viability of returning marine turtles to the functional population after they strand, from a management and asset efficacy perspective, these known prevalences may allow the selection of resources that favour the rescue and treatment of juvenile green marine turtles during winter stranding near one of the identified hotspots. Further, this gave a baseline of environmental conditions that result in elevated stranding rates from which to create predictive models.

8.3.3. Objective 3: Examine the causes of animals being sent to rehabilitation and the outcome for each specific one (Chapter 4)

Anecdotal evidence prior to the commencement of this study suggested the majority of animals rehabilitated were often being found either re-stranded with the same initial cause of stranding and within a short time or dead. However, we showed different causes of stranding influenced the survival for individuals, in terms of length of time in care, and survival in rehabilitation and post rehabilitation success. Anthropogenic causes of stranding were most successfully treated whereas animals showing signs of disease and buoyancy disorders were least successfully treated. Accordingly, if the goal of rehabilitation is solely to return as many marine turtles as possible to the functional population, rehabilitation centres could focus their resources and attention to marine turtles which are more likely to survive rehabilitation and return to the ocean. Based on our study, this would suggest efforts should be directed to anthropogenic causes of stranding and identifying the unknown causes of stranding. This should not mean other causes of stranding are not treated, but it does give insight into the contribution of rehabilitation to the wild population. On an animal level, rehabilitation offers the opportunity to improve the health and well-being of individuals with a secondary focus on determining the causes of unknown strandings to better contribute to our body of knowledge on reason associated with stranding. Unfortunately, unknown causes of stranding are still the predominant recorded reason for stranding and may influence future strategies as we fill this knowledge gap.

8.3.4. Objective 4: Examine the link between stranded turtles and their input into key wild populations (Chapter 4 and 5)

Despite the challenges of rehabilitation, a large number of animals were released over an 18 year period leading up to 2013. However very few of them (2%) were recaptured as a successful part of the healthy population, despite high healthy population recapture rates between 8%-84.3% depending on age class (Bell et al., 2012; Chaloupka and Limpus, 2005, 2002). This low rate of returning animals to the functional population and high operating costs of many rehabilitation facilities can be crudely expressed as a cost of $123,000 per turtle per successful return to the wild. Further supporting the notion to selectively rehabilitate, was that of the 5491 turtles found stranded alive during this 18-

year investigation, 2052 of them were released *in situ* with only minimal triage and assessment. Of these 2052, 429 were recaptured as part of the healthy functional population. Superficially, this suggests rehabilitation may not be a cost-effective strategy for marine turtle population recovery plans in Queensland coastal waters. However, there were limitations to this analysis which included, among other things, the severity of impediment. In *in situ* triaged animals, the cause of stranding was likely minor whereas it was likely life threatening to warrant transportation to a rehabilitation facility. However, even if rehabilitation does not serve a direct population benefit, it has other applications which benefit marine turtle wild populations such as increasing public awareness as to the implications of their actions and environmental behaviours (conservation education), the benefit to the welfare and prevention of suffering of individuals and the contribution to veterinary medicine through diagnostic and pathological discovery from unsuccessfully rehabilitated individuals. These reasons should be considered when looking at the alternatives or better management practices.

8.3.5. Objective 5: Help management agencies to designate appropriate rehabilitation facilities (Chapter 4)

Even though the number of dead turtles that strand is only an index on the actual number of animal which die in total (Epperly et al., 1996; Peltier et al., 2012), monitoring stranding of marine turtles along the coastline provides a cost-effective powerful first line tool in gathering data to make management decisions. By extension, the service rehabilitation facilities perform through diagnosing and identifying causes of impediment is equally valuable. Often rehabilitation centres are run from donations and contributions of volunteers. The offset of expenses such as diagnostics has helped gain a financially and scientifically valuable insight into the challenges facing marine turtles, and provide a greater depth of informed knowledge to those charged with managing our natural coastal resources and endangered species populations.

Based on the cost of rehabilitation at the three main rehabilitation centres in Queensland it may not be economically viable to treat all marine turtles that strand given the low return rate to the functional wild population. However using the stranding data and the cause of strandings examined throughout this study, there may be alternative options that consider the welfare and prevention of suffering of individuals as well as the economic realities of rehabilitation. These approaches warrant further investigation.

The creation of MASH (Mobile Army Surgical Hospital) response units may be an appropriate strategy. These units focus on *in situ* triage and treatment. Being equipped to respond to a range of known (common- identified throughout this study) conditions for specific species and age classes of animals, MASH units may deploy to stranding outbreaks or temporarily set up camp seasonally to respond to increased strandings that are known to occur in certain areas based on trends and cycles. This latter use of these cost saving units can now be employed with a greater understanding of the effect of extreme weather on stranding and with the capacity to predict where strandings will occur ahead of time.

8.3.6. Objective 6: Increase the level of understanding about the implications that extreme weather events cause to marine turtles (Chapter 6)

It had been suggested that marine turtle stranding numbers are affected by weather events. However there has been no definitive link established or time frame determined between the weather event and the increase in stranding numbers.

By modelling known weather events and known stranding rates for numerous locations along the Queensland coastline over an 18-year period for which both datasets were collected, we determined that strandings occurred after a lag phase, with increased water discharge having the greatest effect on stranding numbers. It was found that the marine turtle stranding rate increased 7-12 months after an increase in water discharge levels. Increased water discharge is likely to occur as the four major catchment basins of Queensland fill and release water into the ocean. Flooding or prolonged rain events are the most common causes of this event. It was also found that mean monthly maximum, mean monthly minimum and average daily temperature change also affected marine turtle stranding rates, suggesting increased changes in ambient temperature above the normal range is not tolerated well by marine turtle populations. In its most extreme form, it is common for marine turtles to strand *en masse* if water temperatures rapidly change by 10°C or more in a short period of time. Known as cold stunning, this phenomenon periodically occurs in regions such as the southern United States and, based on the patterns seen in this study, may be present in a subtler form along the Queensland coastline. An associated disease syndrome, cold stress, has been demonstrated to occur with dugongs in Queensland (Owen et al., 2013). Both cold stunning and cold stress are known to occur in turtles and sirenia, respectively, in the same regions of the United States.

Knowing the weather patterns and these relationships, it is now possible for marine resource managers to more effectively respond to marine turtle strandings if they can incorporate these singular models of known effector and response into a comprehensive model that predicts the cumulative impact of extraordinary weather events.

8.3.7. Objective 7: Allow the prediction of stranding rates following extreme weather events to allow facilities to better respond to increases in stranded animals (Chapter 7)

Building on the singular lag phase models, this study developed the first predictive models for assessing the impact of adverse weather conditions and catastrophic events on green marine turtle strandings, using the largest long term dataset currently available. These models used empirical data to support the hypothesis by Marsh and Kwan (2008), Meager and Limpus (2012), Flint et al. (2015) and Flint et al. (submitted) that there is a lag phase of catastrophe to stranding based on the severity of the catastrophe to demonstrate there is a predictable link between weather events and stranding rates.

By combining the individual effects of discharge, temperature change and rainfall, we created the template for a model that can assist those charged with responding to strandings in a number of ways. First responders are able to not only know the number of strandings to expect under average conditions, but this model produced the minimum number of strandings to expect under maximum adverse environmental conditions and the maximum number of strandings to expect under minimum adverse environmental conditions. With current long range weather forecasting capacities, this allows management teams to budget resources for disasters well in advance of the actual need. Further, this model can identify which species and age class are likely to need treatment and when. Combining this with known disease prevalence for each of these cohorts, cost effective response teams, such as MASH units, can be stocked and deployed to optimise their efforts by knowing what they will encounter, how many they will encounter and where they will encounter them.

8.3.8. Objective 8: Through the increased understanding of their effects, determine the net benefit of rehabilitation and better predict and prepare for the effects of future extreme weather events.

It is difficult to assess the true success of rehabilitation without following each individual to determine what its fate is, whether it really have been treated, whether it succumbed to the original cause of stranding again or whether it went on thrive and participate in the normal healthy functional population. This study offered the first study looking at marine turtles at a population level in Queensland and their survival post release from rehabilitation. Superficially, it appears that rehabilitation is an expensive practice that does not significantly contribute to the overall health of the functional population. The number of turtles reported stranded during this study represent approximately >0.001% ((9641 turtles/18 years)/641262) of the suspected benthic southern Great Barrier Reef population (Chaloupka, 2002b). When analysing the number of animals which are rehabilitated this number gets exponentially smaller, indicating on a population level that rehabilitation is not a cost effective conservation tool.

Although the success of individuals was small, this does not negate the role of rehabilitation or iterations of rehabilitation along the Queensland coastline. The creation of MASH units which can be deployed to help triage and treat turtles has been used when responding the cold stun events in southern United States as well as in other species, such as dolphins and sirenia.

8.3.9. Objective 9: Develop methods for consideration by management agencies to recognise and respond to mass/unusual mortality or disease events appropriately (Chapter 6)

Management agencies are often charged with the impossible- conserving and restoring a species with little to no budget in addition to regulating all of those involved in trying to assist. As marine turtles are seen as sentinel indicators of ecosystem health (Aguirre and Lutz, 2004) and as flagship species for conservation (Tisdell and Wilson, 2003) their survival is important for a variety reasons to protect ecological, aesthetic, economic, existence and bequest values (Aguirre and Lutz, 2004; Chaloupka et al., 2008b; Feck and Hamann, 2013; Jackson et al., 2001; Tisdell and Wilson, 2001).

A uniform set of management guidelines for each species in question is often required to harmoniously advance our understanding of the stressors facing a particular species and optimise the efforts of all of those involved. Recovery Plans such as those for marine turtles in Australia and the United States are one such example of a living document that may benefit from these concerted efforts.

The presented patterns, known environmental stressors and created models can be used to target when, where, and who are stranding along the Queensland coastline. We still need to further determine why these animals are stranding. Once we have this piece of the puzzle, we will be in a better position to decide if rehabilitation is required, and in what form. Until then, rehabilitation centres may need to shift their focus to the preferential treatment of the known (when, where and who) and partner with clinical and anatomic pathologist to determine the why (unknown) if they are to remain a contributing component of marine turtle survivorship.

8.4. Limitations

There were several limitations to this overall study.

The number of unknown causes of stranding and general missing pieces of information resulted in a large loss of usable data. For examples, outcome of rehabilitation, date of outcome and release information. However, this served as a benchmark for areas in which we know we must improve, specifically to allow an improved, more robust predictive model to be developed.

As with any exploratory modelling, we identified several limitations that may influence the accuracy of any resultant outputs including distributed sample equality, equal adequate sample sizes for each species, availability of additional environmental data such as seagrass abundance, habitat type and the distance from shore that an event was recorded. This may be addressed by the additional of other variables into the equation.

One of the limitations of these models is that the stranding sample size was different for each examined latitudinal block. Larger sample size may make the relationships more noticeable than smaller sample sizes, but as this used one of the longest running and largest datasets available, this may be difficult to correct. The modelling aspects of this study, may also have been strengthened by determining if different species showed different responses times and directions. This was not possible due to the small sample sizes of the other species occurring within the study location. Small sample sizes may be overcome with time by continued support of this long term program to ensure ongoing collection of data. The accuracy of the prediction produced by the model may be improved using small sample sizes by data transformation techniques or including other variables to verify estimates.

The models may have been strengthened by the use of food availability/viability as an additional factor. However, due to the paucity of seagrass and other aquatic vegetation abundance data, it was decided to use weather as a proxy as weather data is available in immediate time. Further, there is evidence that discharge and rainfall are adequate proxies for seagrass abundance as large-scale seagrass die-off have been closely associated in time and intensity to flooding (Poiner et al., 1993b; Preen et al., 1995; Wetz and Yoskowitz, 2013). For future studies, comparing these three factors to pick the best variable or combination of weighted variables is required.

These models were also limited by only analysing the highest value QAIC value and not using the full range of lag periods. This method was selected as it uses the lag periods where the most variation is mathematically explained by the model, but this approach has the potential to create problems in that although the QAIC value was the highest, there was not always a strong significant relationship between these variables. Potential solutions for this may be trialling the model with different lag times to see if the results are different.

During the analysis of this data, it has become apparent that coding methodology was not the same through the years. When this study was undertaken and the two databases (QTCP database and StrandNet) married together, in order to gather as many stranding records as possible the QTCP database was interrogated for animals which had been reported dead, that needed rescue or were sent to rehabilitation. Unfortunately, it appears that some animals within that database which were encountered during the Shark Control Program were not noted as needing rescue, and as such had not been included under the stranded category. This oversight did not affect the model creation as only unknown and natural causes of death were used. However, it does potentially affect some small reporting components of Chapter 3 and 4. A quick analysis *a posteriori* revealed that the proportions were still the same in terms of animals which stranded and died vs stranded alive, so is not believed to impact any data interpretation or conclusions drawn. The complete dataset was used for Chapter 5.

Although MASH units on the surface may appear to be beneficial, there needs to be more research and identification of on-costs associated with them. For example, costs associated with having staff on-call, staff location allowance and the storage and maintenance of equipment for extended periods of time. Although many institutions already have staff paid to be on-call and pay staff location allowance when they respond to strandings, this may add extra costs.

8.5. Future Studies

Although providing a solid platform from which to estimate future strandings, one of the immediate benefits of the predictive model is that is has enabled us to identify gaps in our current datasets and understanding of the interplay between the examined environmental variables. Further work needs to be done to investigate the feasibility of using food availability as a variable. Also, further investigation into the model to enable it to better predict the stranding numbers so that is can be used to more accurately determine marine turtle stranding rates in the future and be able to be used in multiple locations both nationally and internationally.

Once the predictive model has been refined, investigations can continue in order to provide better resources to first responders and marine resource managers. This may be in the form of protocols for better strategies to optimally rehabilitate; for example, MASH units which can be deployed to different locations depending on the need. If developed as light weight mobile units in the ilk of those used by the Army Medical Corp, these units also have the potential to be placed in remote areas or in areas where it is difficult to move animals from. This may enable better care to be given to animals that strand outside of the traditional populated areas or classic accessible beach front.

Further investigation is also required into the currently unknown causes of strandings; which account for a large proportion of all recorded stranding. The identification of these causes may improve the success of rehabilitation and allow animals which strand due to this reason to have a better outcome. This may come in the form of more detailed diagnostic procedures on living animals or more detailed necropsies on dead animals to gain a better understanding of underlying causes and their prevalence. Both would benefit from involving clinical and anatomical pathologist expertise to facilitate diagnoses. This will create a greater depth of knowledge that allows first responders to identify similar cases in the future.

This study has investigated the trends of marine turtle strandings and has furthered the knowledge of some of the causative agents and their interplay with a range of environmental variables. Further investigation of these causes will improve marine turtle survivorship post rehabilitation. Further investigation into the effects of environmental variables and how they can be used to more accurately predict marine turtle stranding numbers in the future will enable marine turtle resource managers to be better prepared and provide better care to animals which have stranded.

References

Abal, E.G., Dennison, W.C., 1996. Seagrass depth range and water quality in southern Moreton Bay, Queensland, Australia. Mar. Freshw. Res. 47, 763–771. doi:http://dx.doi.org/10.1071/MF9960763

Addison, D.S., Nelson, K.A., 2000. Recapture of a Tagged, Captive Reared Juvenile Loggerhead Turtle - An Example of Habituation? Mar. Turt. Newsl. 89, 15–16.

Adelman, L.M., Falk, J.H., James, S., 2000. Impact of National Aquarium in Baltimore on Visitors' Conservation Attitudes, Behavior, and Knowledge. Curator Museum J. 43, 33–61. doi:10.1111/j.2151-6952.2000.tb01158.x

Aguirre, A.A., Lutz, P., 2004. Marine Turtles as Sentinels of Ecosystem Health: Is Fibropapillomatosis an Indicator? Ecohealth 1, 275–283. doi:10.1007/s10393-004-0097-3

Aguirre, A.A., Spraker, T.R., Balazs, G.H., Zimmerman, B., 1998. Spirorchidiasis and Fibropapillomatosis in Green Turtles from the Hawaiian Islands. J. Wildl. Dis. 34, 91–98. doi:10.7589/0090-3558-34.1.91

Alfaro-Shigueto, J., Mangel, J.C., Bernedo, F., Dutton, P.H., Seminoff, J.A., Godley, B.J., 2011. Small-scale fisheries of Peru: a major sink for marine turtles in the Pacific. J. Appl. Ecol. 48, 1432–1440. doi:10.1111/j.1365-2664.2011.02040.x

Altwegg, R., Crawford, R.J.M., Underhill, L.G., Williams, A.J., 2008. Long-term survival of de-oiled Cape gannets *Morus capensis* after the Castillo de Bellver oil spill of 1983. Biol. Conserv. 141, 1924–1929. doi:http://dx.doi.org/10.1016/j.biocon.2008.04.030

Anderson, D.R., 2008. Model Based Inference in the Life Sciences: A Primer on Evidence. Springer New York, New York, NY. doi:10.1007/978-0-387-74075-1

Anderson, D.W., Gress, F., Michael Fry, D., 1996. Survival and dispersal of oiled brown pelicans after rehabilitation and release. Mar. Pollut. Bull. 32, 711–718. doi:10.1016/0025-326X(96)00027-6

Arthur, K.E., McMahon, K., Limpus, C.J., Dennison, W.C., 2009. Feeding Ecology of Green Turtles (Chelonia mydas) from Shoalwater Bay, Australia. Mar. Turt. Newsl. 123, 6–12.

Avens, L., Braun-McNeill, J., Epperly, S.P., Lohmann, K.J., 2003. Site fidelity and homing behavior in juvenile loggerhead sea turtles (Caretta caretta). Mar. Biol. 143, 211–220. doi:10.1007/s00227-003-1085-9

Baker, L., Edwards, W., Pike, D.A., 2015. Sea turtle rehabilitation success increases with body size and differs among species. Endanger. Species Res. 29, 13–21. doi:10.3354/esr00696

Balazs, G.H., 1991. Current status of fibropapillomas in the Hawaiian green turtle, *Chelonia mydas*, in: Balazs, G.H., Pooley, S.G. (Eds.), Research Plan for Marine Turtle Fibropapilloma. NOAA Technical Memorandum, Honolulu, Hawaii, pp. 47–58.

Ballantyne, R., Packer, J., Hughes, K., Dierking, L., 2007. Conservation learning in wildlife tourism settings: lessons from research in zoos and aquariums. Environ. Educ. Res. 13, 367–383. doi:10.1080/13504620701430604

Barlow, A.M., Mullineaux, E., Wood, R., Taweenan, W., Wastling, J.M., 2010. Giardiosis in Eurasian badgers (*Meles meles*). Vet. Rec. 167, 1017. doi:10.1136/vr.c7346

Barlow, A.M., Schock, A., Bradshaw, J., Mullineaux, E., Dastjerdi, A., Everest, D.J., McGowan, S., Steinbach, F., Cowen, S., 2012. Parvovirus enteritis in Eurasian badgers (*Meles meles*). Vet. Rec. 170, 416–416. doi:10.1136/vr.100680

Bell, I.P., Schwarzkopf, L., Manicom, C., 2012. High survivorship of an annually decreasing aggregation of hawksbill turtles, *Eretmochelys imbricata*, found foraging in the northern Great Barrier Reef. Aquat. Conserv. Mar. Freshw. Ecosyst. 22, 673–682. doi:10.1002/aqc.2245

Bell, J., Lovelock, C.E., 2013. Insuring Mangrove Forests for Their Role in Mitigating Coastal Erosion and Storm -Surge: An Australian Case Study. Wetlands 33, 279–289. doi:10.1007/s13157-013-0382-4

Bellido, J.J., Báez, J.C., Castillo, J.J., Martín, J.J., Mons, J.L., Real, R., 2010. Unusual behaviour of an immature loggerhead turtle released in the Alboran Sea. Anim. Biodivers. Conserv. Biol. 1, 15–18.

Bettinger, J.M., Bettoli, P.W., 2002. Fate, Dispersal, and Persistence of Recently Stocked and Resident Rainbow Trout in a Tennessee Tailwater. North Am. J. Fish. Manag. 22, 425–432. doi:10.1577/1548-8675(2002)022<0425:FDAPOR>2.0.CO;2

Bhaskaran, K., Gasparrini, A., Hajat, S., Smeeth, L., Armstrong, B., 2013. Time series regression studies in environmental epidemiology. Int. J. Epidemiol. 42, 1187–95. doi:10.1093/ije/dyt092

Biddle, T.., Limpus, C.J., 2011. Marine wildlife stranding and mortality database annual reports 2005 – 2010. Marine turtles, Conservation Technical Data Report 2010 (1). 1-124. Brisbane, Queensland.

Bjorndal, K.A., 1997. Foraging ecology and nutrition of sea turtles. Biol. sea turtles 1, 199–

231.

Bolker, B., 2017. Dealing with quasi-models in R.

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., White, J.S.S., 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends Ecol. Evol. 24, 127–135. doi:10.1016/j.tree.2008.10.008

Bolker, B.M., R Development Core Team, 2016. bbmle: Tools for General Maximum Likelihood Estimation.

Bosagna, R.G., 2012. The Rehabilitation and Release of Marine Fauna in South East Queensland: Green Sea Turtles (*Chelonia mydas*) as a case study. The Univeristy of Queensland, Gatton, Queensland.

Boschat, G., Pezza, A., Simmonds, I., Perkins, S., Cowan, T., Purich, A., 2015. Large scale and sub-regional connections in the lead up to summer heat wave and extreme rainfall events in eastern Australia. Clim. Dyn. 44, 1823–1840. doi:10.1007/s00382-014-2214-5

Bowen, K.D., Spencer, R.-J., Janzen, F.J., 2005. A comparative study of environmental factors that affect nesting in Australian and North American freshwater turtles. J. Zool. 267, 397. doi:10.1017/S0952836905007533

Brand-Gardner, S.J., Lanyon, J.M., Limpus, C.J., 1999. Diet selection by immature green turtles, *Chelonia mydas*, in subtropical Moreton Bay, south-east Queensland. Aust. J. Zool. 47, 181. doi:10.1071/ZO98065

Broderick, A.C., Coyne, M.S., Fuller, W.J., Glen, F., Godley, B.J., 2007. Fidelity and over-wintering of sea turtles. Proc. Biol. Sci. 274, 1533–1538. doi:10.1098/rspb.2007.0211

Broderick, A.C., Godley, B.J.B.J., Hays, G.C., Godley, B.J.B.J., 2001. Trophic status drives interannual variability in nesting numbers of marine turtles. Proc. R. Soc. B Biol. Sci. 268, 1481–1487. doi:10.1098/rspb.2001.1695

Brodie, J.E., Ariel, E., Thomas, C., Brien, O., Berry, K., 2014. Links between water quality and marine turtle health 88.

Brodie, J.E., Devlin, M.J., Haynes, D., Waterhouse, J., 2011. Assessment of the eutrophication status of the Great Barrier Reef lagoon (Australia). Biogeochemistry 106, 281–302. doi:10.1007/s10533-010-9542-2

Brodie, J.E., Kroon, F.J., Schaffelke, B., Wolanski, E.C., Lewis, S.E., Devlin, M.J., Bohnet, I.C., Bainbridge, Z.T., Waterhouse, J., Davis, M., 2012. Terrestrial pollutant runoff to the Great Barrier Reef: An update of issues, priorities and management responses.

Mar. Pollut. Bull. 65, 81–100. doi:http://dx.doi.org/10.1016/j.marpolbul.2011.12.012

Brodie, J.E., Mitchell, A.W., 2005. Nutrients in Australian tropical rivers: Changes with agricultural development and implications for receiving environments, in: Marine and Freshwater Research. pp. 279–302. doi:10.1071/MF04081

Brodie, J.E., Waterhouse, J., 2012. A critical review of environmental management of the “not so Great” Barrier Reef. Estuar. Coast. Shelf Sci. 104–105, 1–22. doi:10.1016/j.ecss.2012.03.012

Burnham, K.P., Anderson, D.R., 2004. Multimodel Inference: Understanding AIC and BIC in Model Selection. Sociol. Methods Res. 33, 261–304. doi:10.1177/0049124104268644

Burnham, K.P., Anderson, D.R., 2002. Model selection and multimodel inference, Second Edi. ed, NY: Springer. Springer, New York.

Caillouet Jr, C.W., Duronslet, M.J., Landry Jr, A.M., Revera, D.B., Shaver, D.J., Stanley, K.M., Heinly, R.W., Stabenau, E.K., 1991. Sea turtle strandings and shrimp fishing effort in the northwestern Gulf of Mexico, 1986-89. Fish. Bull. 89, 712–718.

Calleson, C.., 2014. Issues and opportunities associated with using manatee mortality data to evaluate the effectiveness of manatee protection efforts in Florida. Endanger. Species Res. 26, 127–136. doi:10.3354/esr00638

Calleson, C., Frohlich, R., 2007. Slower boat speeds reduce risks to manatees. Endanger. Species Res. 3, 295–304. doi:10.3354/esr00056

Campbell, L.M., 2003. Contemporary Culture,Use and Conservation of Sea Turtles, in: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles Vol II. CRC Press, Boca Raton, pp. 307–338.

Campbell, S.J., McKenzie, L.J., 2004. Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. Estuar. Coast. Shelf Sci. 60, 477–490. doi:http://dx.doi.org/10.1016/j.ecss.2004.02.007

Cardona, L., Fernández, G., Revelles, M., Aguilar, A., 2012. Readaptation to the wild of rehabilitated loggerhead sea turtles (*Caretta caretta*) assessed by satellite telemetry. Aquat. Conserv. Mar. Freshw. Ecosyst. 22, 104–112. doi:10.1002/aqc.1242

Cardoso, P.G., Raffaelli, D., Pardal, M.A., 2008. The impact of extreme weather events on the seagrass Zostera noltii and related Hydrobia ulvae population. Mar. Pollut. Bull. 56, 483–492. doi:http://dx.doi.org/10.1016/j.marpolbul.2007.11.006

Carr, A., Caldwell, D.K., 1956. The Ecology and Migrations of Sea Turtles, 1. Results of

Field Work in Florida, 1955. Am. Museum Novit. 1–24.

Chaloupka, M., 2002a. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. Ecol. Modell. 148, 79–109. doi:10.1016/s0304-3800(01)00433-1

Chaloupka, M., 2002b. Phase 1 - Assessment of suitability of Queensland Parks & Wildlife Service Sea Turtle Data for use in models of the population dynamics of the Southern Great Barrier Reef Turtle Stock. Gt. Barrier Reef Mar. Park Auth. Res. Publ. 74, 1–42. doi:10.1016/S0304-3800(01)00433-1

Chaloupka, M., Balazs, G.H., Work, T.M., 2009. Rise and Fall over 26 years of a Marine Epizootic in Hawaiian Green Sea Turtles. J. Wildl. Dis. 45, 1138–1142. doi:10.2307/20094599

Chaloupka, M., Bjorndal, K.A., Balazs, G.H., Bolten, A.B., Ehrhart, L.M., Limpus, C.J., Suganuma, H., Troëng, S., Yamaguchi, M., 2008a. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Glob. Ecol. Biogeogr. 17, 297–304. doi:10.1111/j.1466-8238.2007.00367.x

Chaloupka, M., Limpus, C.J., 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Mar. Biol. 146, 1251–1261. doi:10.1007/s00227-004-1512-6

Chaloupka, M., Limpus, C.J., 2002. Survival probability estimates for the endangered loggerhead sea turtle resident in southern Great Barrier Reef waters. Mar. Biol. 140, 267–277. doi:10.1007/s002270100697

Chaloupka, M., Limpus, C.J., 2001. Trends in the abundance of sea turtles resident in southern Great Barrier Reef waters. Biol. Conserv. 102, 235–249. doi:http://dx.doi.org/10.1016/S0006-3207(01)00106-9

Chaloupka, M., Work, T.M., Balazs, G.H., Murakawa, S.K.K., Morris, R., 2008b. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). Mar. Biol. 154, 887–898. doi:10.1007/s00227-008-0981-4

Chandler, R.B., Andrew Royle, J., 2013. Spatially explicit models for inference about density in unmarked or partially marked populations. Ann. Appl. Stat. 7, 936–954. doi:10.1214/12-AOAS610

Chapman, P.A., 2017. Diversity, impacts and diagnosis of pathogenic parasites in green sea turtles (*Chelonia mydas*) from Queensland, Australia. University of Queensland.

Chapman, P.A., Cribb, T.H., Blair, D., Traub, R.J., Kyaw-Tanner, M.T., Flint, M., Mills, P.C., 2015. Molecular analysis of the genera *Hapalotrema* Looss, 1899 and *Learedius* Price, 1934 (Digenea: Spirorchiidae) reveals potential cryptic species, with comments on the validity of the genus *Learedius*. Syst. Parasitol. 90, 67–79. doi:10.1007/s11230-014-9535-y

Chapman, P.A., Owen, H., Flint, M., Traub, R.J., Cribb, T.H., Mills, P.C., 2016. Molecular Characterization of Coccidia Associated with an Epizootic in Green Sea Turtles (*Chelonia mydas*) in South East Queensland, Australia. PLoS One 11, e0149962. doi:10.1371/journal.pone.0149962

Cheng, I., Huang, C., Hung, P., Ke, B., Kuo, C., 2009. Ten Years of Monitoring the Nesting Ecology of the Green Turtle,. Zool. Stud. 48, 83–94.

Coghlan, A., 2014. A Little Book of R for Time Series. Wellcome Trust Sanger Institure, Cambridge, UK.

Coles, R., Grech, A., Rasheed, M., McKenzie, L.J., 2012. Evaluating risk to seagrasses in the Tropical Indo-Pacific Region.

Coles, R., McKenzie, L.J., Rasheed, M.A., Mellors, J., Taylor, H.A., Dew, K., McKenna, S., Sankey, T., Carter, A.B., Grech, A., 2007. Status and trends of seagrass habitats in the Great Barrier Reef World Heritage Area. Department of Primary Industries and Fisheries.

Coles, R., Rasheed, M.A., McKenzie, L.J., Grech, A., York, P.H., Sheaves, M., McKenna, S., Bryant, C., 2015. The Great Barrier Reef World Heritage Area seagrasses: Managing this iconic Australian ecosystem resource for the future. Estuar. Coast. Shelf Sci. 153, A1–A12. doi:10.1016/j.ecss.2014.07.020

Cooper, J., Cooper, M.E., 2006. Ethical and legal implications of treating casualty wild animals. In Pract. 28, 2–6. doi:10.1136/inpract.28.1.2

Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260. doi:10.1038/387253a0

Crow, J.M., 2011. Queensland floods hit Great Barrier Reef. Nature. doi:10.1038/news.2011.21

Cuttriss, A.K., Prince, J.B., Castley, J.G., 2013. Seagrass communities in southern Moreton Bay, Australia: Coverage and fragmentation trends between 1987 and 2005. Aquat. Bot. 108, 41–47. doi:http://dx.doi.org/10.1016/j.aquabot.2013.03.003

Daunt, F., Afanasyev, V., Silk, J.R.D., Wanless, S., 2006. Extrinsic and intrinsic determinants of winter foraging and breeding phenology in a temperate seabird. Behav. Ecol. Sociobiol. 59, 381–388. doi:10.1007/s00265-005-0061-4

Day, J.C., Dobbs, K., 2013. Effective governance of a large and complex cross-jurisdictional marine protected area: Australia’s Great Barrier Reef. Mar. Policy 41, 14–24. doi:http://dx.doi.org/10.1016/j.marpol.2012.12.020

Department of Environment and Heritage Protection, 1992. Code of Practice Code of Practice – Care of sick, injured or orphaned protected animals in Queensland. Nature Conservation Act 1992. Department of Environment and Heritage Protection, Brisbane.

Devlin, M.J., Schaffelke, B., 2009. Spatial extent of riverine flood plumes and exposure of marine ecosystems in the Tully coastal region, Great Barrier Reef. Mar. Freshw. Res. 60, 1109–1122. doi:http://dx.doi.org/10.1071/MF08343

Devlin, M.J., Wenger, A.S., Da Silva, E., Álvarez-Romero, J.G., Waterhouse, J., McKenzie, L.J., 2012a. Extreme weather conditions in the Great Barrier Reef: Drivers of change?, in: Yellowlees, D., Hughes, T.P. (Eds.), 12th International Coral Reef Symposium. James Cook Univer, Cairns, Australia.

Devlin, M.J., Wenger, A.S., Petus, C., da Silva, E.T., Debose, J., Álvarez-Romero, J.G., 2012b. Reef Rescue Marine Monitoring Program. Final Report of JCU Activities 2011/12 - Flood Plumes and Extreme weather monitoring for the Great Barrier Reef Marine Park Authority. James Cook University, Townsville.

Dobbs, K., 2001. Marine Turtles in the Great Barrier Reef World Heritage Area: a compendium of information and basis for the development of policies and strategies for the conservation of marine turtles., 1st Editio. ed. Great Barrier Reef Marine Park Authority, Townsville, Qld.

Doherty, P.F., White, G.C., Burnham, K.P., 2012. Comparison of model building and selection strategies. J. Ornithol. 152, 317–323. doi:10.1007/s10336-010-0598-5

Doody, J.S., Georges, A., Young, J.E., 2003. Twice every second year: reproduction in the pig-nosed turtle, Carettochelys insculpta, in the wet-dry tropics of Australia. J. Zool. 259, 179–188. doi:10.1017/S0952836902003217

Dubois, S., 2003. A survey of wildlife rehabilitation goals, impediments, issues and success in British Columbia, Canada 101. doi:10.1024/0301-1526.32.1.54

Easterling, D.R., Evans, J.L., Groisman, P.Y., Karl, T.R., Kunkel, K.E., Ambenje, P., 2000a. Observed Variability and Trends in Extreme Climate Events: A Brief Review*. Bull. Am. Meteorol. Soc. 81, 417–425. doi:10.1175/1520-0477(2000)081<0417:OVATIE>2.3.CO;2

Easterling, D.R., Meehl, G.A., Parmesan, C., Changnon, S.A., Karl, T.R., Mearns, L.O., 2000b. Climate Extremes: Observations, Modeling, and Impacts. Science (80-.). 289, 2068–2074. doi:10.1126/science.289.5487.2068

Ebner, B.C., Thiem, J.D., 2009. Monitoring by telemetry reveals differences in movement and survival following hatchery or wild rearing of an endangered fish. Mar. Freshw. Res. 60, 45–57. doi:http://dx.doi.org/10.1071/MF08027

Environment Australia, 2003. Recovery Plan for Marine Turtles in Australia. doi:0 642 21436 0

Environmental Protection Agency, 2008. Indo-Pacific Marine Turtles Identification Guide.

Epperly, S.P., Braun-McNeill, J., Chester, A.J., Cross, F.A., Merriner, J. V., Tester, P.A., 1995. Winter distribution of sea turtles in the vicinity of Cape Hatteras and their interactions with the summer flounder trawl fishery. Bull. Mar. Sci. 56, 547–568.

Epperly, S.P., Braun-McNeill, J., Chester, A.J., Cross, F.A., Merriner, J. V., Tester, P.A., Churchill, J.H., Braun, J., Chester, A.J., Cross, F.A., Merriner, J. V., Tester, P.A., Churchill, J.H., 1996. Beach strandings as an indicator of at-sea mortality of sea turtles. Bull. Mar. Sci. 59, 289–297. doi:10.1017/CBO9781107415324.004

Estes, J.A., 1998. Concerns about Rehabilitation of Oiled Wildlife Concerns about Rehabilitation of Oiled Wildlife. Conserv. Biol. 12, 1156–1157.

Fabricius, K.E., 2005. Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. Mar. Pollut. Bull. 50, 125–146. doi:10.1016/j.marpolbul.2004.11.028

Fabricius, K.E., 2005. No Title, Marine Pollution Bulletin. doi:10.1016/j.marpolbul.2004.11.028

Fabricius, K.E., De 'ath, G., Humphrey, C., Zagorskis, I., Schaffelke, B., 2012. Intra-annual variation in turbidity in response to terrestrial runoff on near-shore coral reefs of the Great Barrier Reef. doi:10.1016/j.ecss.2012.03.010

Fajardo, I., Babiloni, G., Miranda, Y., 2000. Rehabilitated and wild barn owls *(Tyto alba)*: Dispersal, life expectancy and mortality in Spain. Biol. Conserv. 94, 287–295. doi:10.1016/S0006-3207(00)00003-3

Falk, J.H., Reinhard, E.M., Vernon, C.L., Bronnenkant, K., Heimlich, J.E., Deans, N.L.,

2007. Why Zoos & Aquariums Matter : Assessing the Impact of a Visit to a Zoo or Aquarium. Assoc. Zoos Aquariums 24, 1–24. doi:10.1306/D4269792-2B26-11D7-8648000102C1865D

Feck, A.D., Hamann, M., 2013. Effect of sea turtle rehabilitation centres in Queensland, Australia, on people's perceptions of conservation. Endanger. Species Res. 20, 153–165. doi:10.3354/esr00482

Fleming, I.A., Gross, M.R., 1993. Breeding Success of Hatchery and Wild Coho Salmon (*Oncorhynchus kisutch*) in Competition. Ecol. Appl. 3, 230–245. doi:10.2307/1941826

Flint, J., Flint, M., Limpus, C.J., Mills, P.C., 2017a. Status of Marine Turtle Rehabilitation in Queensland. PeerJ Accepted, e3132. doi:10.7717/peerj.3132

Flint, J., Flint, M., Limpus, C.J., Mills, P.C., 2017b. The impact of environmental factors on stranding rates. PLoS One A.

Flint, J., Flint, M., Limpus, C.J., Mills, P.C., 2015. Trends in Marine Turtle Strandings along the East Queensland, Australia Coast, between 1996 and 2013. J. Mar. Biol. 2015, 1–7. doi:10.1155/2015/848923

Flint, M., 2013. Free-Ranging Sea Turtle Health, in: Wyneken, J., Lohmann, K.J., Musick, J.A. (Eds.), The Biology of Sea Turtles , Volume III. CRC Press, Boca Raton, pp. 379–397.

Flint, M., 2010. An assessment of disease on the health of green (*Chelonia mydas*) and loggerhead (*Caretta caretta*) turtles in southern Queensland Australia. The Univeristy of Queensland.

Flint, M., Blair, D., Patterson-Kane, J.C., Kyaw-Tanner, M., Mills, P.C., 2010a. Blood flukes (Spirorchiidae) as a major cause of marine turtle mortality in Queensland. XII Int. Congr. Parasitol. .

Flint, M., Eden, P.A., Limpus, C.J., Owen, H., Gaus, C., Mills, P.C., 2014. Clinical and Pathological Findings in Green Turtles (*Chelonia mydas*) from Gladstone, Queensland: Investigations of a Stranding Epidemic. Ecohealth 12, 298–309. doi:10.1007/s10393-014-0972-5

Flint, M., Limpus, C.J., Patterson-Kane, J.C., Murray, P.J., Mills, P.C., 2010b. Corneal Fibropapillomatosis in Green Sea Turtles (*Chelonia mydas*) in Australia. J. Comp. Pathol. 142, 341–6. doi:10.1016/j.jcpa.2009.10.012

Flint, M., Morton, J.M., Limpus, C.J., Patterson-Kane, J.C., Murray, P.J., Mills, P.C., 2010c. Development and application of biochemical and haematological reference intervals to identify unhealthy green sea turtles (*Chelonia mydas*). Vet. J. 185, 299–304. doi:http://dx.doi.org/10.1016/j.tvjl.2009.06.011

Flint, M., Patterson-Kane, J.C., Limpus, C.J., Mills, P.C., 2010d. Health surveillance of stranded green turtles in Southern Queensland, Australia (2006-2009): An epidemiological analysis of causes of disease and mortality. Ecohealth 7, 135–145. doi:10.1007/s10393-010-0300-7

Flint, M., Patterson-Kane, J.C., Limpus, C.J., Work, T.M., Blair, D., Mills, P.C., 2009. Postmortem Diagnostic Investigation of Disease in Free-Ranging Marine Turtle Populations: A Review of Common Pathologic Findings and Protocols. J. Vet. Diagnostic Investig. 21, 733–759. doi:10.1177/104063870902100601

Flint, P.L., Fowler, A.C., 1998. A drift experiment to assess the influence of wind on recovery of oiled seabirds on St Paul Island, Alaska. Mar. Pollut. Bull. 36, 165–166. doi:10.1016/S0025-326X(97)00178-1

Foley, A.M., Bagley, D., Bailey, R., Belskis, L., Brost, B., Evou, N., Hardy, R., Hirama, S., Hoffman, A., Koperski, M., Meylan, A.B., Minch, K., Patrick, L., Provancha, J., Schaf, S., Schroeder, B.A., Stacy, B.A., Teas, W., Trindeli, R., Witherington, B.E., 2012. Report on a massive hypothermic stunning event of sea turtles in Florida during January 2010., in: Proceedings Of The Thirty-First Annual Symposium On Sea Turtle Biology And Conservation. NOAA Technical Memorandum.

Fuentes, M.M.P.B., Hamann, M., Lukoschek, V., 2012. Marine Climate Change in Australia Marine Reptiles 379–400.

Fuentes, M.M.P.B., Limpus, C.J., Hamann, M., 2011. Vulnerability of sea turtle nesting grounds to climate change. Glob. Chang. Biol. 17, 140–153. doi:10.1111/j.1365-2486.2010.02192.x

Furnas, M.J., 2003. Catchments and Corals: Terrestrial Runoff to the Great Barrier Reef.

Gasparrini, A., 2011. Distributed Lag Linear and Non-Linear Models in R: The Package dlnm. J. Stat. Softw. 43, 1–20.

Gaus, C., Grant, S., Jin, N.L., Goot, K., Chen, L., Villa, A., Neugebauer, F., Qi, L., Limpus, C.J., 2012. Investigation of contaminant levels in green turtles from Gladstone.

Geraci, J.R., Lounsbury, V.J., 2005. Marine Mammals Ashore: A Field Guide for Strandings, Second Edi. ed. National Aquarium in Baltimore, Baltimore, MD.

Gibbes, B., Grinham, A., Neil, D., Olds, A., Maxwell, P., Connolly, R., Weber, T., Udy, N., Udy, J., 2014. Moreton Bay and Its Estuaries: A Sub-tropical System Under Pressure from Rapid Population Growth. Springer Netherlands, pp. 203–222. doi:10.1007/978-94-007-7019-5_12

Glazebrook, J.S., Campbell, R.S.F., 1981. Pathological changes associated with cardiovascular trematodes (Digenea: Spirorchidae) in a green sea turtle *Chelonia mydas*. J. Comp. Pathol. 91, 361–368.

Glazebrook, J.S., Campbell, R.S.F., Blair, D., 1989. Studies on cardiovascular fluke (Digenea: Spirorchiidae) infections in sea turtles from the Great Barrier Reef, Queensland, Australia. J. Comp. Pathol. 101, 231–250. doi:10.1016/0021-9975(89)90033-9

Godley, B.J., Blumenthal, J.M., Broderick, A.C., Coyne, M.S., Godfrey, M.H., Hawkes, L.A., Witt, M.J., 2008. Satellite tracking of sea turtles: Where have we been and where do we go next? Endanger. Species Res. 4, 3–22. doi:10.3354/esr00060

Gordon, A.N., 2005. A Necropsy-based Study of Green Turtles (*Chelonia mydas*) in South-East Queensland. The Univeristy of Queensland.

Gordon, A.N., Kelly, W.R., Cribb, T.H., 1998. Lesions Caused by Cardiovascular Flukes (Digenea: Spirorchidae) in Stranded Green Turtles (*Chelonia mydas*). Vet. Pathol. 35, 21–30. doi:10.1177/030098589803500102

Gordon, A.N., Kelly, W.R., Lester, R.J.G., 1993. Epizootic mortality of free-living green turtles, *Chelonia mydas*, due to coccidiosis. J. Wildl. Dis. 29, 490–494.

Graham, M.H., 2003. Confronting Multicollinearity in Ecological Multiple Regression. Ecology 84, 2809–2815. doi:10.1890/02-3114

Great Barrier Reef Marine Park Authority, 2015. Reef 2050 Long-Term Sustainability Plan.

Great Barrier Reef Marine Park Authority, 2014. Great Barrier Reef Outlook Report 2014. Great Barrier Reef Marine Park Authority, Townsville.

Great Barrier Reef Marine Park Authority, 2011a. Extreme Weather and the Great Barrier Reef. Great Barrier Reef Marine Park Authority. doi:978-1-921682-71-1 (pdf)

Great Barrier Reef Marine Park Authority, 2011b. Impacts of tropical cyclone Yasi on the Great Barrier Reef: a report on the findings of a rapid ecological impact assessment, July 2011. Townsville.

Great Barrier Reef Marine Park Authority, 2007. Great Barrier Reef Marine Turtles Identification sheet. Great Barrier Reef Marine Park Authority, Townsville.

Grech, A., Chartrand-Miller, K., Erftemeijer, P., Fonseca, M., McKenzie, L., Rasheed, M., Taylor, H., Coles, R., 2012. A comparison of threats, vulnerabilities and management

approaches in global seagrass bioregions. Environ. Res. Lett. 7, 24006. doi:10.1088/1748-9326/7/2/024006
Grech, A., Coles, R., Marsh, H., 2011. A broad-scale assessment of the risk to coastal seagrasses from cumulative threats. Mar. Policy 35, 560–567. doi:http://dx.doi.org/10.1016/j.marpol.2011.03.003
Greenland, J.A., Limpus, C.J., 2004. Marine wildlife stranding and mortality database annual report 2003. III. Turtles. Conserv. Tech. Data Rep. 2003, 1–25. doi:1449–194X
Grogan, A., Kelly, A., 2013. A review of RSPCA research into wildlife rehabilitation. Vet. Rec. 172, 211. doi:10.1136/vr.101139
Hamann, M., Godfrey, M.H., Seminoff, J.A., Arthur, K.E., Barata, P.C.R., Bjorndal, K.A., Bolten, A.B., Broderick, A.C., Campbell, L.M., Carreras, C., Casale, P., Chaloupka, M., Chan, S.K.F., Coyne, M.S., Crowder, L.B., Diez, C.E., Dutton, P.H., Epperly, S.P., FitzSimmons, N.N., Formia, A., Girondot, M., Hays, G.C., Cheng, I.J.I., Kaska, Y., Lewison, R.L., Mortimer, J.A., Nichols, W.J., Reina, R.D., Shanker, K., Spotila, J.R., Tomás, J., Wallace, B.P., Work, T.M., Zbinden, J., Godley, B.J.B., 2010. Global research priorities for sea turtles: informing management and conservation in the 21st century. Endanger. Species Res. 11, 245–269. doi:10.3354/esr00279
Hamann, M., Limpus, C.J., Owens, D.W., 2003. Reproductive cycles of males and females, in: Lutz L, P., Muzick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles. CRC Press, Boca Raton, pp. 135–161.
Hamann, M., Limpus, C.J., Whittier, J.M., 2002. Patterns of lipid storage and mobilisation in the female green sea turtle (*Chelonia mydas*). J. Comp. Physiol. B 172, 485–493. doi:10.1007/s00360-002-0271-2
Haward, M., Davidson, J., Lockwood, M., Hockings, M., Kriwoken, L., Allchin, R., 2013. Climate change, scenarios and marine biodiversity conservation. Mar. Policy 38, 438–446. doi:http://dx.doi.org/10.1016/j.marpol.2012.07.004
Hawkes, L.A., Broderick, A.C., Godfrey, M.H., Godley, B.J., 2009. Climate change and marine turtles. Endanger. SPECIES Res. Endang Species Res 7, 137–154. doi:10.3354/esr00198
Hawkes, L.A., Broderick, A.C., Godfrey, M.H., Godley, B.J., 2007. Investigating the potential impacts of climate change on a marine turtle population. Glob. Chang. Biol. 13, 923–932. doi:10.1111/j.1365-2486.2007.01320.x
Hawkes, L.A., Tomás, J., Revuelta, O., León, Y., Blumenthal, J., Broderick, A., Fish, M.,

Raga, J., Witt, M., Godley, B., 2012. Migratory patterns in hawksbill turtles described by satellite tracking. Mar. Ecol. Prog. Ser. 461, 223–232. doi:10.3354/meps09778

Hays, G.C., 2014. Tracking animals to their death. J. Anim. Ecol. 83, 5–6. doi:10.1111/1365-2656.12164

Hazel, J., Gyuris, E., 2006. Vessel-related mortality of sea turtles in Queensland, Australia. Wildl. Res. 33, 149–154. doi:10.1071/WR04097

Hazel, J., Lawler, I.R., Hamann, M., 2009. Diving at the shallow end: Green turtle behaviour in near-shore foraging habitat. J. Exp. Mar. Bio. Ecol. 371, 84–92. doi:10.1016/j.jembe.2009.01.007

Heck, K.L., Carruthers, T.J.B., Duarte, C.M., Randall Hughes, A., Kendrick, G., Orth, R.J., Williams, S.W., 2008. Trophic transfers from seagrass meadows subsidize diverse marine and terrestrial consumers. Ecosystems. doi:10.1007/s10021-008-9155-y

Herbst, L.H., 1994. Fibropapillomatosis of Marine Turtles. Annu. Rev. Fish Dis. 4, 389–425. doi:10.1016/0959-8030(94)90037-X

Herbst, L.H., Jacobson, E.R., Klein, P.A., Balazs, G.H., Moretti, R., Brown, T., Sundberg, J.P., 1999. Comparative pathology and pathogenesis of spontaneous and experimentally induced fibropapillomas of green turtles (*Chelonia mydas*). Vet. Pathol. 36, 551–564. doi:10.1354/vp.36-6-551

Herr, A., Greiner, R., Stoeckl, N., 2004. Understanding Adoption of On-farm Conservation Practices in the Burdekin Dry Tropics, Queensland. Australas. J. Environ. Manag. 11, 278–288. doi:10.1080/14486563.2004.10648624

Hillestad, H., Richardson, J.I., Williamson, G., 1978. Incidental capture of sea turtles by shrimp trawlermen in Georgia. Proc. Ann. Conf. S.E. Assoc. Fish Wild. Agencies 167–178.

Hirayama, R., 1998. Oldest known sea turtle. Nature 392, 705–708. doi:http://www.nature.com/nature/journal/v392/n6677/suppinfo/392705a0_S1.html

Howes, M., Dodson, J., Tomerini, D., 2013. Planning for resilience in a changing climate: Integrating spatial analysis and on-line pollution inventories to manage chemical releases during floods, in: Planning for Resilient Cities and Regions. AESOP & ACSP.

Hoyos, N., Escobar, J., Restrepo, J.C., Arango, A.M., Ortiz, J.C., 2013. Impact of the 2010–2011 La Niña phenomenon in Colombia, South America: The human toll of an extreme weather event. Appl. Geogr. 39, 16–25. doi:http://dx.doi.org/10.1016/j.apgeog.2012.11.018

Hyland, S.J., Courtney, A.J., Butler, C.T., 1989. Distribution of seagrass in the Moreton region from Coolangatta to Noosa. .

Jackson, J.B., Kirby, M.X., Berger, W.H., Bjorndal, K. a, Botsford, L.W., Bourque, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J. a, Hughes, T.P., Kidwell, S., Lange, C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J., Warner, R.R., 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293, 629–37. doi:10.1126/science.1059199

Jensen, M.P., Fitzsimmons, N.N., Dutton, P.H., 2013. Molecular Genetic of Sea Turtles, in: The Biology of Sea Turtles , Volume III. CRC Press, Boca Raton.

Jessop, T.S., Hamann, M., Limpus, C.J., 2004. Body condition and physiological changes in male green turtles during breeding. Mar. Ecol. Prog. Ser. 276, 281–288. doi:10.3354/meps276281

Joo, M., Raymond, M.A.A., McNeil, V.H., Huggins, R., Turner, R.D.R., Choy, S., 2012. Estimates of sediment and nutrient loads in 10 major catchments draining to the Great Barrier Reef during 2006–2009. Mar. Pollut. Bull. 65, 150–166. doi:http://dx.doi.org/10.1016/j.marpolbul.2012.01.002

Karesh, W.B., 1995. Wildlife Rehabilitation : Additional Considerations for Developing Countries. J. Zoo Wildl. Med. 26, 2–9.

Kerr, E.A., Strother, S., 1990. Seasonal changes in standing crop of *Zostera muelleri* in south-eastern Australia. Aquat. Bot. 38, 369–376. doi:10.1016/0304-3770(90)90031-F

Kéry, M., Schaub, M., 2012. Bayesian population analysis using WinBUGS: a hierarchical perspective. Academic Press.

Kirkman, H., 1978. Decline of seagrass in northern areas of Moreton Bay, Queensland. Aquat. Bot. 5, 63–76. doi:10.1016/0304-3770(78)90047-5

Kirkwood, J.K., 2003. Introduction: wildlife casualties and the veterinary surgeon., in: Mullineaux, E., Best, D., Cooper, J.. (Eds.), BSAVA Manual of Wildlife Casualties. BSAVA Publications, Gloucester, MA, USA, pp. 1–5.

Komoroske, L.M., Jensen, M.P., Stewart, K.R., Shamblin, B.M., Dutton, P.H., 2017. Advances in the Application of Genetics in Marine Turtle Biology and Conservation. Front. Mar. Sci. 4, 156. doi:10.3389/fmars.2017.00156

Kwan, D., 1994. Fat reserves and reproduction in the green turtle, *Chelonia mydas* . Wildl. Res. 21, 257–265. doi:http://dx.doi.org/10.1071/WR9940257

Lamont, M.M., Fujisaki, I., 2014. Effects of Ocean Temperature on Nesting Phenology and

Fecundity of the Loggerhead Sea Turtle (Caretta caretta). J. Herpetol. 48, 98–102. doi:10.1670/12-217

Landsberg, J.H., Balazs, G.H., Steidinger, K.A., Baden, D.G., Work, T.M., Russell, D.J., 1999. The potential role of natural tumor promoters in marine turtle fibropapillomatosis. J. Aquat. Anim. Health 11, 199–210. doi:10.1577/1548-8667(1999)011<0199

Lanyon, J.M., Marsh, H., 1995. Temporal changes in the abundance of some tropical intertidal seagrasses in North Queensland. Aquat. Bot. 49, 217–237. doi:10.1016/0304-3770(94)00435-O

Leeney, R.H.R.R.H., Amies, R., Broderick, A.C., Witt, M.J., Loveridge, J., Doyle, J., Godley, B.J., 2008. Spatio-temporal analysis of cetacean strandings and bycatch in a UK fisheries hotspot. Biodivers. Conserv. 17, 2323–2338. doi:10.1007/s10531-008-9377-5

Lewison, R.L., Crowder, L.B., Read, A.J., Freeman, S.A., 2004a. Understanding impacts of fisheries bycatch on marine megafauna. Trends Ecol. Evol. 19, 598–604. doi:http://dx.doi.org/10.1016/j.tree.2004.09.004

Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S., DiMatteo, A.D., Dunn, D.C., Kot, C.Y., Bjorkland, R.K., Kelez, S., Soykan, C., Stewart, K.R., Sims, M., Boustany, A., Read, A.J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proc. Natl. Acad. Sci. 111, 5271–5276. doi:10.1073/pnas.1318960111

Lewison, R.L., Freeman, S.A., Crowder, L.B., 2004b. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecol. Lett. 7, 221–231. doi:10.1111/j.1461-0248.2004.00573.x

Limpus, C.J., 2009. A Biological Review Of Australian Marine Turtles. 6. Leatherback Turtle, *Dermochelys coriacea* (Vandelli). Environmental Protection Agency, Brisbane.

Limpus, C.J., 2008a. A Biological Review of Australian Marine Turtles. 2. Green Turtle, *Chelonia mydas* (Linnaeus). Environmental Protection Agency, Brisbane.

Limpus, C.J., 2008b. A biological review of Australian marine turtles. 1. Loggerhead Turtle *Caretta caretta* (Linnaeus). Environmental Protection Agency.

Limpus, C.J., 2008c. A biological review of Australian marine turtle species, 4. Olive ridley turtle Lepidochelys olivacea (Eschscholtz). Environmental Protection Agency.

Limpus, C.J., 2007. A biological review of Australian marine turtles. 5. Flatback Turtle Natator depresses (Garman). Environmental Protection Agency.

Limpus, C.J., 1993. The green turtle, *Chelonia mydas*, in Queensland: breeding males in the southern Great Barrier Reef. Wildl. Res. 20, 513–523. doi:http://dx.doi.org/10.1071/WR9930513

Limpus, C.J., 1992. The hawksbill turtle, *Eretmochelys imbricata*, in Queensland: population structure within a southern Great Barrier Reef feeding ground. Wildl. Res. 19, 489–505. doi:10.1071/WR9920489

Limpus, C.J., 1989. Environmental regulation of green turtle breeding in eastern Australia., in: Eckert, S.A., Eckert, K.L., Richardson, Thelma, H. (Eds.), Processing of the Ninth Annual Workshop on Sea Turtle Conservation and Biology. U.S. Department of Commerce, National Oceanographic and Atmospheric Administration, Jekyll Island, Georgia, pp. 97–98.

Limpus, C.J., 1985. A study of the loggerhead sea turtle, Caretta caretta, in Eastern Australia. University of Queensland, Brisbane, Queensland.

Limpus, C.J., 1978. The reef: uncertain land of plenty. Explor. North, A Nat. Hist. Queensland. Richmond Hill Press. Melb. 187–222.

Limpus, C.J., Chaloupka, M., 1997. Nonparametric regression modelling of green sea turtle growth rates (southern Great Barrier Reef). Mar. Ecol. Prog. Ser. 149, 23–34. doi:10.3354/meps149023

Limpus, C.J., Couper, P.J., Read, M.A., 1994a. The Green Turtle, *Chelonia mydas*, in Queensland: Population structure in a Warm Temperate feeding area. Mem. Queensl. Museum. Brisbane 35, 139–154.

Limpus, C.J., Couper, P.J., Read, M.A., 1994b. The loggerhead turtle, *Caretta caretta*, in Queensland: population structure in a warm temperate feeding area. Mem. Queensl. Museum 37, 195–204. doi:10.1016/B978-1-4160-4389-8.50161-8

Limpus, C.J., de Villiers, D.L., de Villiers, M.A., Limpus, D.J., Read, M.A., 2001. The loggerhead turtle, *Caretta caretta*, in Queensland: observations on feeding ecology in warm temperate waters. Mem. Queensl. Museum 46, 631–645.

Limpus, C.J., Limpus, D.J., 2003. Biology of the loggerhead turtle in western South Pacific Ocean foraging areas, in: Bolten, A.B., Witherington, B.E. (Eds.), Loggerhead Sea Turtles. Smithsonian Institution, pp. 93–113.

Limpus, C.J., Limpus, D.J., 2001. The loggerhead turtle, *Caretta caretta*, in Queensland:

breeding migrations and fidelity to a warm temperate feeding area. Chelonian Conserv. Biol. 4, 142–153.

Limpus, C.J., Limpus, D.J., Arthur, K.E., Parmenter, C.J., 2005. Monitoring green turtle population dynamics in Shoalwater Bay: 2000 - 2004, Great Barrier Reef Marine Park Authority Research Publication. Great Barrier Reef Marine Park Authority, Townsville.

Limpus, C.J., Limpus, D.J., Munchow, M., Barnes, P., 2007. Queensland Turtle Conservation Project Raine Island Turtle Study, 2004-2005. Environ. Prot. 2005, 41.

Limpus, C.J., Limpus, D.J., Savige, M., Shearer, D., 2011. Health Assessment of Green Turtles in South and Central Queensland Following Extreme Weather Impacts on Coastal Habitat During 2011, Conservation Technical and Data Report 2011(4).

Limpus, C.J., Miller, J.D., Parmenter, C.J., Limpus, D.J., 2003. The green turtle, *Chelonia mydas*, population of Raine Island and the northern Great Barrier Reef: 1843-2001. Mem. MUSEUM 49, 349–440.

Limpus, C.J., Miller, J.D., Parmenter, C.J., Reimer, D., McLachlan, N., Webb, R., 1992. Migration of green (*Chelonia mydas*) and loggerhead (*Caretta caretta*) turtles to and from eastern Australian rookeries. Wildl. Res. 19, 347–357. doi:http://dx.doi.org/10.1071/WR9920347

Limpus, C.J., Nicholls, N., 2000. ENSO regulation of Indo-pacific green turtle populations. Aust. Exp. 21, 399–408. doi:10.1007/978-94-015-9351-9_24

Limpus, C.J., Nicholls, N., 1988. The Southern Oscillation Regulates the Annual Numbers of Green Turtles (*Chelonia mydas*) Breeding Around Northern Australia. Wildl. Res. 15, 157–161. doi:http://dx.doi.org/10.1071/WR9880157

Limpus, C.J., Reed, P.C., 1985a. The green turtle, *Chelonia mydas*, in Queensland: a preliminary description of the population structure in a coral reef feeding ground, in: Grigg, G., Shine, R., Ehmann, H. (Eds.), Biology of Australasian Frogs and Reptiles. Surrey Beatty and Sons, Sydney, pp. 47–52.

Limpus, C.J., Reed, P.C., 1985b. Green sea turtles stranded by Cyclone Kathy on the south-western coast of the Gulf of Carpentaria. Aust. Wildl. Res. 12, 523–533.

Limpus, C.J., Reed, P.C., Miller, J.D., 1985. Temperature Dependent Sex Determination in Queensland Sea Turtle: Intraspecific Variation in Caretta caretta., in: Grigg, G.C., Shine, R., Ehmann, H. (Eds.), Biology of Australasian Frogs and Reptiles. Routledge, pp. 343–351.

Limpus, C.J., Reimer, D., 1990. The loggerhead turtle, *Caretta caretta* in Queensland: a population in decline Proceedings of the Australian Marine Turtle Conservation Workshop, 14--17 November 1990, Sea World Nara Resort, Gold Coast, Australia. Queensl. Dep. Environ. Herit. Aust. Conserv. Agency, Brisbane, Aust. 39–59.

Link, W.A., Barker, R.J., 2006. Model weights and the foundations of multimodel inference. Ecology 87, 2626–2635. doi:10.1890/0012-9658(2006)87[2626:MWATFO]2.0.CO;2

Lloyd, H.B., Ross, G.A., 2015. Long-term trends in cetacean incidents in New South Wales, Australia. Aust. Zool. 37, 492–500. doi:10.7882/AZ.2015.015

Loftin, R.W., 1985. The Medical Treatment of Wild Animals. Environ. Ethics 7, 231–239. doi:10.5840/enviroethics19857339

Long, B., Skewes, T., Taranto, T., Thomas, M., Isdale, P., Pitcher, R., Poiner, I., 1997. Seagrass dieback in north western Torres Strait. Tech. Report, CSIRO Mar. Res. Clevel.

Marcovaldi, M., Lopez, G., Soares, L., Lima, E., Thomé, J., Almeida, A., 2010. Satellite-tracking of female loggerhead turtles highlights fidelity behavior in northeastern Brazil. Endanger. Species Res. 12, 263–272. doi:10.3354/esr00308

Marsh, H., Freeland, W.J., Limpus, C.J., Reed, P.C., 1986. The Stranding of Dugong and Sea Turtles resulting from Cyclone Kathy, March 1984: A Report on the Rescue Effort and Biological Data Obtained. Conservation Comission of the Northern Territory.

Marsh, H., Kwan, D., 2008. Temporal variability in the life history and reproductive biology of female dugongs in Torres Strait: The likely role of sea grass dieback. Cont. Shelf Res. 28, 2152–2159. doi:10.1016/j.csr.2008.03.023

Martin, J., Edwards, H.H., Fonnesbeck, C.J., Koslovsky, S.M., Harmak, C.W., Dane, T.M., 2015. Combining information for monitoring at large spatial scales: First statewide abundance estimate of the Florida manatee. Biol. Conserv. 186, 44–51. doi:10.1016/j.biocon.2015.02.029

Mazzoil, M.S., McCulloch, S.D., Youngbluth, M.J., Kilpatrick, D.S., Murdoch, M.E., Mase-Guthrie, B., Odell, D.K., Bossart, G.D., 2008. Radio-Tracking and Survivorship of Two Rehabilitated Bottlenose Dolphins (*Tursiops truncatus*) in the Indian River Lagoon, Florida. Aquat. Mamm. 34, 54–64.

McBride, J.L., Haylock, M.R., Nicholls, N., 2003. Relationships between the Maritime Continent Heat Source and the El Niño–Southern Oscillation Phenomenon. J. Clim. 16, 2905–2914. doi:10.1175/1520-0442(2003)016<2905:RBTMCH>2.0.CO;2

McBride, J.L., Nicholls, N., 1983. Seasonal Relationships between Australian Rainfall and the Southern Oscillation. Mon. Weather Rev. 111, 1998–2004. doi:10.1175/1520-0493(1983)111<1998:SRBARA>2.0.CO;2

McFee, W.E., Hopkins-Murphy, S.R.., Schwacke, L.H., 2006. Trends in bottlenose dolphin (*Tursiops truncatus*) strandings in South Carolina , USA , 1997-2003 : implications for the Southern North Carolina and South Carolina Management Units. J. cetacean Res. Manag. 8, 195–201.

McKenzie, L.J., Collier, C., Waycott, M., 2014. Reef Rescue Marine Monitoring Program - Inshore Seagrass, Annual Report for the sampling period 1st July 2011 – 31st May 2012. James Cook University, Cairns.

McKenzie, L.J., Collier, C., Waycott, M., Unsworth, R.K.F., Yoshida, R.L., Smith, N., 2012. Monitoring inshore seagrasses of the GBR and responses to water quality. Symposium 9–13.

McKenzie, L.J., Mellors, J., Waycott, M., Udy, J.W., Coles, R., 2010. 6. Intertidal Seagrass Monitoring 237–285.

McKenzie, L.J., Roder, C.A., Roelofs, A.J., Lee Long, W.J., 2000. Post-flood Monitoring of Seagrasses in Hervey Bay and the Great Sandy Strait 1999: Implications for Dugong, Turtle and Fisheries Management. DPI Inf. Ser. 46pp.

McLeod, A.I., Yu, H., Mahdi, E., 2011. Time Series Analysis with R. Handb. Stat. 30, 1–23. doi:http://dx.doi.org/10.1016/B978-0-444-53858-1.00023-5

Meager, J.J., Limpus, C.J., 2014. Mortality of Inshore Marine Mammals in Eastern Australia Is Predicted by Freshwater Discharge and Air Temperature. PLoS One 9, e94849. doi:10.1371/journal.pone.0094849

Meager, J.J., Limpus, C.J., 2012a. Marine wildlife stranding and mortality database annual report 2011. I. Dugong. Conserv. Tech. Data Rep., Conservation Technical and Data Report 2012(3) 1, 1–46. doi:1449–194X

Meager, J.J., Limpus, C.J., 2012b. Marine wildlife stranding and mortality database annual report 2011 III. Marine Turtles. Brisbane, Queensland.

Meager, J.J., Sumpton, W.D., 2016. Bycatch and strandings programs as ecological indicators for data-limited cetaceans. Ecol. Indic. 60, 987–995. doi:10.1016/j.ecolind.2015.08.052

Meehl, G.A., Karl, T., Easterling, D.R., Changnon, S.A., Pielke, R., Changnon, D., Evans, J., Groisman, P.Y., Knutson, T.R., Kunkel, K.E., Mearns, L.O., Parmesan, C., Pulwarty, R., Root, T., Sylves, R.T., Whetton, P., Zwiers, F.W., 2000a. An Introduction to Trends in Extreme Weather and Climate Events: Observations, Socioeconomic Impacts, Terrestrial Ecological Impacts, and Model Projections*. Bull. Am. Meteorol. Soc. 81, 413–416. doi:10.1175/1520-0477(2000)081<0413:AITTIE>2.3.CO;2

Meehl, G.A., Zwiers, F., Evans, J., Knutson, T., Mearns, L., Whetton, P., 2000b. Trends in Extreme Weather and Climate Events: Issues Related to Modeling Extremes in Projections of Future Climate Change. Bull. Am. Meteorol. Soc. 81, 427–436. doi:http://dx.doi.org/10.1175/1520-0477(2000)081<0427:TIEWAC>2.3.CO;2

Mellors, J., Marsh, H., Coles, R., 1993. Intra-annual changes in seagrass standing crop, Green Island, Northern Queensland. Mar. Freshw. Res. 44, 33–41. doi:10.1071/MF9930033

Mestre, F., Bragança, M.P., Nunes, A., dos Santos, M.E., 2014. Satellite tracking of sea turtles released after prolonged captivity periods. Mar. Biol. Res. 10, 996–1006. doi:10.1080/17451000.2013.872801

Moein, S., Lenhardt, M., Barnard, D., Keinath, J., Musick, J., 1993. Marine turtle auditory behavior. J. Acoust. Soc. Am. 93, 2378–2378. doi:10.1121/1.406096

Moein Bartol, S., Musick, J.A., 2003. Sensory Biology of Sea Turtles, in: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles Vol II. CRC Press, Boca Raton, pp. 79–102.

Moein Bartol, S., Musick, J.A., Lenhardt, M.L., 1999. Auditory Evoked Potentials of the Loggerhead Sea Turtle (Caretta caretta). Source: Copeia 836–840.

Moore, M., Early, G., Touhey, K., Barco, S.G., Gulland, F., Wells, R.S., 2007. Rehabilitation and release of marine mammals in the United States: Risks and benefits. Mar. Mammal Sci. 23, 731–750. doi:10.1111/j.1748-7692.2007.00146.x

Mullineaux, E., 2014. Veterinary treatment and rehabilitation of indigenous wildlife. J. Small Anim. Pract. 55, 293–300. doi:10.1111/jsap.12213

Murphy, T.M., Hopkins-Murphy, S.R.R., 1989. Sea turtle and shrimp fishing interactions: a summary and critique of relevant information. Cent. Mar. Conserv. Washington, D.C. 52 p.

Murray, K., Conner, M.M., 2009. Methods to quantify variable importance: implications for the analysis of noisy ecological data. Ecology 90, 348–355.

Musick, J.A., Limpus, C.J., Lutz, P.L., Musick, J.A., 1997. Habitat utilization and migration in juvenile sea turtles, in: The Biology of Sea Turtles. CTC Marine science series, pp. 137–163.

National Research Council Committee on Sea Turtle Conservation, 1990. Decline of the Sea Turtles: Causes and Prevention. National Academies Press. doi:10.1016/0160-4120(92)90079-J

Nawojchik, R., Aubin, D.J.S., Johnson, A., Aquarium, M., Boulevard, C., Aquarium, M., Aquarium, E., Aquarium, N.E., Wharf, C., Nawojchik, R., Aubin, D.J.S., Johnson, A., 2003. Movements and dive behavior of two stranded, rehabilitated long-finned pilot whales (*Globicephala melas*) in the Northwest Atlantic. Mar. Mammal Sci. 19, 232–239. doi:10.1111/j.1748-7692.2003.tb01105.x

Nicholls, N., Alexander, L. V, 2007. Has the climate become more variable or extreme? Progress 1992-2006. Prog. Phys. Geogr. 31, 77–87. doi:10.1177/0309133307073885

Nicholls, N., Drosdowsky, W., Lavery, B., 1997. Australian rainfall variability and change. Weather 52, 66–72. doi:10.1002/j.1477-8696.1997.tb06274.x

Nichols, W.J., Resendiz, A., Seminoff, J.A., Resendiz, B., 2000. Transpacific Migration of a Loggerhead Turtle Monitored by Satellite Telemetry. Bull. Mar. Sci. 67, 937–947.

Norman, S.A., Huggins, J., Carpenter, T.E., Case, J.T., Lambourn, D.M., Rice, J., Calambokidis, J., Gaydos, J.K., Hanson, M.B., Duffield, D.A., Dubpernell, S., Berta, S., Klope, M., 2012. The application of GIS and spatiotemporal analyses to investigations of unusual marine mammal strandings and mortality events. Mar. Mammal Sci. 28, 251–266. doi:10.1111/j.1748-7692.2011.00507.x

Norton, T.M., Jacobson, E.R., Sundberg, J.P., 1990. Cutaneous Fibropapillomas and Renal Myxofibroma in a Green Turtle, Chelonia mydas. Source J. Wildl. Dis. Wildl. Dis. Assoc. J. Wildl. Dis. 26, 265–270. doi:10.7589/0090-3558-26.2.265

Ogden, J.C., Robinson, L., Whitlock, K., Daganhardt, H., Cebula, R., 1983. Diel foraging patterns in juvenile green turtles (*Chelonia mydas* L.) in St. Croix United States virgin islands. J. Exp. Mar. Bio. Ecol. 66, 199–205. doi:10.1016/0022-0981(83)90160-0

Olds, A.D., Pitt, K.A., Maxwell, P.S., Babcock, R.C., Rissik, D., Connolly, R.M., 2014. Marine reserves help coastal ecosystems cope with extreme weather. Glob. Chang. Biol. 20, n/a-n/a. doi:10.1111/gcb.12606

Owen, H.C., Flint, M., Limpus, C.J., Palmieri, C., Mills, P.C., 2013. Evidence of sirenian cold stress syndrome in dugongs (Dugong dugon) from southeast Queensland, Australia. Dis. Aquat. Organ. 103, 1–7. doi:10.3354/dao02568

Peckham, S.H., Diaz, D.M., Walli, A., Ruiz, G., Crowder, L.B., Nichols, W.J., 2007. Small-Scale Fisheries Bycatch Jeopardizes Endangered Pacific Loggerhead Turtles. PLoS

One 2. doi:10.1371/journal.pone.0001041

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., Ridoux, V., 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indic. 18, 278–290. doi:10.1016/j.ecolind.2011.11.014

Poiner, I.R., Conacher, C., Loneragan, N., Kenyon, R., Somers, I., 1993a. Effects of cyclones on seagrass communities and penaeid prawn stocks of the Gulf of Carpentaria: Final Report. .

Poiner, I.R., Conacher, C., Loneragan, N.R., Kenyon, R., Sonters, I., 1993b. Effects of Cyclones on seagrass Communities and penaeid prawn stocks of the Gulf of Carpentaria. CSIRO Mar. Lab. Rep. 5, 143–154.

Polidoro, B.A., Carpenter, K.E., Collins, L., Duke, N.C., Ellison, A.M., Ellison, J.C., Farnsworth, E.J., Fernando, E.S., Kathiresan, K., Koedam, N.E., Livingstone, S.R., Miyagi, T., Moore, G.E., Ngoc Nam, V., Ong, J.E., Primavera, J.H., Salmo, S.G., Sanciangco, J.C., Sukardjo, S., Wang, Y., Yong, J.W.H., 2010. The Loss of Species: Mangrove Extinction Risk and Geographic Areas of Global Concern. PLoS One 5, e10095. doi:10.1371/journal.pone.0010095

Polovina, J., Uchida, I., Balazs, G.H., Howell, E.A., Parker, D., Dutton, P.H., 2006. The Kuroshio Extension Bifurcation Region: A pelagic hotspot for juvenile loggerhead sea turtles. Deep Sea Res. Part II Top. Stud. Oceanogr. 53, 326–339. doi:http://dx.doi.org/10.1016/j.dsr2.2006.01.006

Preen, A., 1995. Impacts of dugong foraging on seagrass habitats: observational and experimental evidence for cultivation grazing. Mar. Ecol. Prog. Ser. 124, 201–213. doi:10.3354/meps124201

Preen, A., Marsh, H., 1995. Response of dugongs to large-scale loss of seagrass from Hervey Bay, Queensland Australia. Wildl. Res. 22, 507–519. doi:10.1071/WR9950507

Preen, A.R., Lee Long, W.J., Coles, R., 1995. Flood and cyclone related loss, and partial recovery, of more than 1000 km2 of seagrass in Hervey Bay, Queensland, Australia. Aquat. Bot. 52, 3–17. doi:http://dx.doi.org/10.1016/0304-3770(95)00491-H

Quakenbush, L., Beckmen, K., Brower, C.D.N., 2009. Rehabilitation and release of marine mammals in the United States: Concerns from Alaska. Mar. Mammal Sci. 25, 994–999. doi:10.1111/j.1748-7692.2009.00283.x

R Core Team, 2016. R: A language and environment for statistical computing. R

Foundation for Statistical Computing, Vienna, Austria.

Rasheed, M.A., McKenna, S.A., Carter, A.B., Coles, R., 2014. Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. Mar. Pollut. Bull. 83, 491–499. doi:http://dx.doi.org/10.1016/j.marpolbul.2014.02.013

Rasmusson, E.M., Carpenter, T.H., 1982. Variations in Tropical Sea Surface Temperature and Surface Wind Fields Associated with the Southern Oscillation/El Niño. Mon. Weather Rev. 110, 354–384. doi:10.1175/1520-0493(1982)110<0354:VITSST>2.0.CO;2

Read, M.A., Grigg, G.C., Limpus, C.J., 1996. Body Temperatures and Winter Feeding in Immature Green Turtles, *Chelonia mydas*, in Moreton Bay, Southeastern Queensland. J. Herpetol. 30, 262–265. doi:10.2307/1565520

Read, M.A., Limpus, C.J., 2002. The green turtle, *Chelonia mydas* , in Queensland: Feeding ecology of immature turtles in Moreton Bay, Southeastern Queensland.

Renaud, M., Gitschlag, G., Klima, E., Shah, A., Koi, D., Nance, J., 1991. Evaluation of the Impacts of Turtle Excluder Devices (TEDs) on Shrimp Catch Rates in Coastal Waters of the United States along the Guld of Mexico and Atlantic, September 1989 through August 1990, NOAA Technical Memorandum NMFS-SEFC-288.

Saran, K., Parker, G., Parker, R., Dickman, C.R., 2011. Rehabilitation as a conservation tool: A case study using the common wombat. Pacific Conserv. Biol. 17, 310–319.

Sasso, C.R., Epperly, S.P., 2007. Survival of pelagic juvenile Loggerhead turtles in the open ocean. J. Wildl. Manage. 71, 1830–1835. doi:10.2193/2006-448

Sato, K., Matsuzawa, Y., Tanaka, H., Bando, T., Minamikawa, S., Sakamoto, W., Naito, Y., 1998. Internesting intervals for loggerhead turtles, Caretta caretta, and green turtles, Chelonia mydas, are affected by temperature. Can. J. Zool. 76, 1651–1662. doi:10.1139/z98-107

Schaffelke, B., Mellors, J., Duke, N.C., 2005. Water quality in the Great Barrier Reef region: Responses of mangrove, seagrass and macroalgal communities, in: Marine Pollution Bulletin. pp. 279–296. doi:10.1016/j.marpolbul.2004.10.025

Scherer, A.L., Leal Valls, F.C., Basler, A.B., Scherer, J. d M., Petry, M.V., de Fátima Martins Scherer, J., Petry, M.V., 2014. Life Stages, Anthropogenic Impact, and Temporal Variation of Stranded Sea Turtles in Southern Brazil. Chelonian Conserv. Biol. 13, 42–48. doi:10.2744/CCB-1029.1

Schofield, T.D., Early, G., Wenzel, F.W., Matassa, K., Perry, C., Beekman, G., Whitaker, B., Gebhard, E., Walton, W., Swingle, M.W., 2008. Rehabilitation and Homing Behavior of a Satellite-Tracked Harbor Porpoise (*Phocoena phocoena*). Aquat. Mamm. 34, 1–8.

Schwabe, C.W., 1969. Veterinary Medicine and Human Health. Williams and Wilkins Company, Baltimore, MD, USA.

Seedang, S., Fernald, A.G., Adams, R.M., Landers, D.H., 2008. Economic analysis of water temperature reduction practices in a large river floodplain: an exploratory study of the Willamette River, Oregon. River Res. Appl. 24, 941–959. doi:10.1002/rra

Seneviratne, S.I., Nicholls, N., Easterling, D.R., Goodess, C., Kanae, S., Kossin, J., Luo, Y., Merengo, J., McInnes, K., Rahimi, M., Reichstein, M., Sorteberg, A., Vera, C., Zhang, X., 2012. Changes in climate extremes and their impacts on the natural physical environment.

SEQ Catchments, 2011. The Moreton Bay Oil Spill, Environmental Restoration Progam. SEQ Catchments, Brisbane, Queensland.

Sexton, O.J., Phillips, C., Bramble, J.E., 1990. The Effects of Temperature and Precipitation on the Breeding Migration of the Spotted Salamander (Ambystoma maculatum). Copeia 3, 781–787. doi:10.2307/1446443

Shaver, D.J., Teas, W., 1999. Stranding and salvage networds, in: Eckert, K.L., Bjorndal, K.A., Abreu-Grobois, F.A., Donnelly, M. (Eds.), Research and Managment Techniques for the Conservation of Sea Turtles1. pp. 152–155.

Shimada, T., 2015. Spatial ecology and conservation of sea turtles in coastal foraging habitat.

Shimada, T., Jones, R., Limpus, C.J., Groom, R., Hamann, M., 2016a. Long-term and seasonal patterns of sea turtle home ranges in warm coastal foraging habitats: implications for conservation. Mar. Ecol. Prog. Ser. 562, 163–179. doi:10.3354/meps11972

Shimada, T., Limpus, C.J., Jones, R., Hazel, J., Groom, R., Hamann, M., 2016b. Sea turtles return home after intentional displacement from coastal foraging areas. Mar. Biol. 163:8, 1–14. doi:10.1007/s00227-015-2771-0

Short, F.T., Coles, R., Fortes, M.D., Victor, S., Salik, M., Isnain, I., Andrew, J., Seno, A., 2014. Monitoring in the Western Pacific region shows evidence of seagrass decline in line with global trends. Mar. Pollut. Bull. 83, 408–416.

doi:http://dx.doi.org/10.1016/j.marpolbul.2014.03.036
Short, F.T., Neckles, H.A., 1999. The effects of global climate change on seagrasses. Aquat. Bot. 63, 169–196. doi:http://dx.doi.org/10.1016/S0304-3770(98)00117-X
Short, F.T., Wyllie-Echeverria, S., 1996. Natural and human-induced disturbance of seagrasses. Environ. Conserv. 23, 17–27. doi:doi:10.1017/S0376892900038212
Shumway, R.H., Stoffer, D.S., 2011. Time Series Analysis and Its Applications With R Examples, 3rd Editio. ed, Media. Springer. doi:10.1007/978-1-4419-7865-3
Sikarskie, J.G., 1992. The Role of Veterinary Medicine in Wildlife Rehabilitation. J. Zoo Wildl. Med. 23, 397–400.
Slater, J., Limpus, C.J., Robins, J., Pantus, F., Chaloupka, M., 1998. Risk Assessment of Sea Turtle Capture in the Queensland East Coast Otter Trawl Fishery. Brisbane, Queensland.
Smith, G.M., Coates, C.W., 1938. Fibro-epithelial growths of the skin in large marine turtles, *Chelonia mydas* (Linnaeus). Zoologica 23, 93–98.
Snoddy, J.E., Southward Williard, A., 2010. Movements and post-release mortality of juvenile sea turtles released from gillnets in the lower Cape Fear River, North Carolina, USA. Endanger. Species Res. 12, 235–247. doi:10.3354/esr00305
Speirs, M., 2002. A study of marine turtle populations at the Julian Rocks Aquatic Reserve, northern New South Wales. . Southern Cross University: Lismore.
Spotila, J.R., O’Connor, M.P., Paladino, F.V., 1997. Thermal Biology, in: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, New York, pp. 297–314.
Standora, E.A., Spotila, J.R., 1985. Temperature dependent sex determination in sea turtles. Copeia 1985, 711–722. doi:10.2307/1444765
Steffen, W., Hughes, L., Karoly, D., 2013. The Critical Decade: extreme weather. The Climate Comission, Department of Industry, Innovation, Climate Change, Science, Research and Tertiary Education.
Steffen, W., Hunter, J., Hughes, L., 2014. Counting the Costs : Climate Change and Coastal.
Swimmer, Y., Arauz, R., McCracken, M., Naughton, L.M., Ballestero, J., Musyl, M., Bigelow, K., Brill, R., 2006. Diving behavior and delayed mortality of olive ridley sea turtles *Lepidochelys olivacea* after their release from longline fishing gear. Mar. Ecol. Prog. Ser. 323, 253–261. doi:10.3354/meps323253
Swingle, M.W., Warmolts, D., Keinath, J., Musick, J.A., 1994. Loggerhead sea turtle head-

started evaluation: captive growth rates and post release movements and behaviour, in: Bjorndal, K.A., Bolten, A.B., Johnson, D.A., Eliazar, P.J. (Eds.), . Miami, FL, pp. 289–292.

Tavares, D.C., Fulgencio de Moura, J., Siciliano, S., Reyes, L., Rebstock, G., Griot, K., 2016. Environmental Predictors of Seabird Wrecks in a Tropical Coastal Area. PLoS One 11, e0168717. doi:10.1371/journal.pone.0168717

Thayer, G.W., Bjorndal, K.A., Ogden, J.C., Williams, S.L., Zieman, J.C., 1984. Role of Larger Herbivores in Seagrass Communities. Estuaries 7, 351. doi:10.2307/1351619

Thayer, G.W., Engel, D.W., Bjorndal, K.A., 1982. Evidence for short-circuiting of the detritus cycle of seagrass beds by the green turtle, *Chelonia mydas* L. J. Exp. Mar. Bio. Ecol. 62, 173–183. doi:10.1016/0022-0981(82)90090-9

Thomas, K., Harvey, J.T., Goldstein, T., Barakos, J., Gulland, F., 2010. Movement, dive behavior, and survival of California sea lions (*Zalophus californianus*) post treatment for domoic acid toxicosis. Mar. Mammal Sci. 26, 36–52. doi:10.1111/j.1748-7692.2009.00314.x

Tisdell, C., Wilson, C., 2005. Perceived Impacts of Ecotourism on Environmental Learning and Conservation: Turtle Watching as a Case Study. Environ. Dev. Sustain. 7, 291–302. doi:10.1007/s10668-004-7619-6

Tisdell, C., Wilson, C., 2003. Does Ecotourism Contribute to Sea Turtle Conservation? Is the Flagship Status of Turtles Advantageous?, Economics, Ecology and the Environment.

Tisdell, C., Wilson, C., 2002. Ecotourism for the survival of sea turtles and other wildlife. Biodivers. Conserv. 11, 1521–1538. doi:10.1023/A:1016833300425

Tisdell, C., Wilson, C., 2001. Wildlife-based tourism and increased support for nature conservation financially and otherwise: evidence from sea turtle ecotourism at Mon Repos. Tour. Econ. 7, 233–249. doi:10.5367/000000001101297847

Tomás, J., Dominici, A., Nannarelli, S., Forni, L., Badillo, F.J., Raga, J.A., 2001. From hook to hook: the odyssey of a loggerhead sea turtle in the Mediterranean. Mar. Turt. Newsl. 92, 13–14.

Tribe, A., Brown, P.R., 2000. The role of wildlife rescue groups in the care and rehabilitation of Australian fauna. Hum. Dimens. Wildl. 5, 69–85. doi:10.1080/10871200009359180

Tribe, A., Nottidge, B., Hanger, J., Kawakami, T., 2005. Measuring the success of wildlife rehabilitation. Natl. Wildl. Rehabil. Counc. 1–14.

Tucker, A.D., Read, M.A., 2001. Frequency of Foraging by Gravid Green Turtles (*Chelonia mydas*) at Raine Island, Great Barrier Reef. J. Herpetol. 35, 500–503. doi:10.2307/1565970

Turton, S.M., 2005. Managing Environmental Impacts of Recreation and Tourism in Rainforests of the Wet Tropics of Queensland World Heritage Area. Geogr. Res. 43, 140–151. doi:10.1111/j.1745-5871.2005.00309.x

Vélez-Rubio, G.M., Estrades, A., Fallabrino, A., Tomás, J., 2013. Marine turtle threats in Uruguayan waters: Insights from 12 years of stranding data. Mar. Biol. 160, 2797–2811. doi:10.1007/s00227-013-2272-y

Wachenfeld, D.R., Oliver, J., Morrissey, J.I., 1998. State of the Great Barrier Reef World Heritage Area 1998. Great Barrier Reef Marine Park Authority.

Wallace, B.P., Heppell, S.S., Lewison, R.L., Kelez, S., Crowder, L.B., 2008. Impacts of fisheries bycatch on loggerhead turtles worldwide inferred from reproductive value analyses. J. Appl. Ecol. 45, ???-??? doi:10.1111/j.1365-2664.2008.01507.x

Wallace, B.P., Lewison, R.L., McDonald, S.L., McDonald, R.K., Kot, C.Y., Kelez, S., Bjorkland, R.K., Finkbeiner, E.M., Helmbrecht, S.S., Crowder, L.B., 2010. Global patterns of marine turtle bycatch. Conserv. Lett. 3, 131–142. doi:10.1111/j.1755-263X.2010.00105.x

Watanabe, K.K.K., Hatase, H., Kinoshita, M., Omuta, K., Bando, T., Kamezaki, N., Sato, K., Matsuzawa, Y., Goto, K., Nakashima, Y., Takeshita, H., Aoyama, J., Tsukamoto, K., 2011. Population structure of the loggerhead turtle *Caretta caretta*, a large marine carnivore that exhibits alternative foraging behaviors. Mar. Ecol. Prog. Ser. 424, 273–283. doi:10.3354/meps08989

Waterhouse, J., Brodie, J.E., Lewis, S., Mitchell, A., 2012. Quantifying the sources of pollutants in the Great Barrier Reef catchments and the relative risk to reef ecosystems. Mar. Pollut. Bull. 65, 394–406. doi:http://dx.doi.org/10.1016/j.marpolbul.2011.09.031

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. 106, 12377–12381. doi:10.1073/pnas.0905620106

Waycott, M., Longstaff, B.J., Mellors, J., 2005. Seagrass population dynamics and water quality in the Great Barrier Reef region: A review and future research directions, in: Marine Pollution Bulletin. pp. 343–350. doi:10.1016/j.marpolbul.2005.01.017

Weber, S.B., Broderick, A.C., Groothuis, T.G.G., Ellick, J., Godley, B.J., Blount, J.D., 2012. Fine-scale thermal adaptation in a green turtle nesting population. Proc. R. Soc. B 279, 1077–1084. doi:10.1098/rspb.2011.1238

Webster, P.J., 2005. Changes in Tropical Cyclone Number, Duration, and Intensity in a Warming Environment. Science (80-.). 309, 1844–1846. doi:10.1126/science.1116448

Wells, R.S., Manire, C.A., Byrd, L., Smith, D.R., Gannon, J.G., Fauquier, D., Mullin, K.D., 2009. Movements and dive patterns of a rehabilitated Risso's dolphin, *Grampus griseus*, in the Gulf of Mexico and Atlantic Ocean. Mar. Mammal Sci. 25, 420–429. doi:10.1111/j.1748-7692.2008.00251.x

Wetz, M.S., Yoskowitz, D.W., 2013. An "extreme" future for estuaries? Effects of extreme climatic events on estuarine water quality and ecology. Mar. Pollut. Bull. 69, 7–18. doi:http://dx.doi.org/10.1016/j.marpolbul.2013.01.020

Widmer, W.., Underwood, A.., 2004. Factors affecting traffic and anchoring patterns of recreational boats in Sydney Harbour, Australia. Landsc. Urban Plan. 66, 173–183. doi:10.1016/S0169-2046(03)00099-9

Wilson, C., Tisdell, C., 1999. Sea Turtles as a Non-consumptive tourism resource in Australia (No. Working Paper No. 32), Economics, Ecology and the Environment.

Witherington, B.E., Bresette, M., Herren, R., 2006. *Chelonia mydas* – green turtle., in: Meylan, P. (Ed.), Biology and Conservation of Florida Turtles. Chelonian Research Foundation, pp. 90–104.

Witherington, B.E., Ehrhart, L.M., 1989. Hypothermic stunning and mortality of marine turtles in the Indian river lagoon system, Florida. Copeia 3, 696–703. doi:10.2307/1445497

Wobeser, G.A., 2007. Disease in wild animals: Investigation and management, Disease in Wild Animals: Investigation and Management. doi:10.1007/978-3-540-48978-8

Wolfaardt, A.C., Williams, A.J., Underhill, L.G., Crawford, R.J.M., Whittington, P.A., 2009. Review of the rescue, rehabilitation and restoration of oiled seabirds in South Africa, especially African penguins *Spheniscus demersus* and Cape gannets *Morus capensis* , 1983–2005. African J. Mar. Sci. 31, 31–54. doi:10.2989/AJMS.2009.31.1.3.774

Work, T.M., Balazs, G.H., Summers, T.M., Hapdei, J.R., Tagarino, A.P., 2015. Causes of mortality in green turtles from Hawaii and the insular Pacific exclusive of fibropapillomatosis. Dis. Aquat. Organ. 115, 103–110. doi:10.3354/dao02890

Wyles, K.J., Pahl, S., White, M., Morris, S., Cracknell, D., Thompson, R.C., 2013. Towards a Marine Mindset: Visiting an Aquarium Can Improve Attitudes and Intentions Regarding Marine Sustainability. Visit. Stud. 16, 95–110. doi:10.1080/10645578.2013.768077

Wyneken, J., 2003. The external morphology, Musculoskeletal System, and Neuro-Anatomy of Sea Turtles., in: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles Vol II. CRC Press, Boca Raton.

Wyneken, J., Lohmann, K.J., Musick, J.A., 2013. The biology of Sea Turtles, Vol 3. CRC Press, Boca Raton.

York, P.H., Smith, T.M., Coles, R., McKenna, S.A., Connolly, R.M., Irving, A.D., Jackson, E.L., McMahon, K., Runcie, J.W., Sherman, C.D.H., Sullivan, B.K., Trevathan-Tackett, S.M., Brodersen, K.E., Carter, A.B., Ewers, C.J., Lavery, P.S., Roelfsema, C.M., Sinclair, E.A., Strydom, S., Tanner, J.E., van Dijk, K., Warry, F.Y., Waycott, M., Whitehead, S., 2016. Identifying knowledge gaps in seagrass research and management: An Australian perspective, Marine Environmental Research. doi:10.1016/j.marenvres.2016.06.006

Zeileis, A., Kleiber, C., Jackman, S., 2008. Regression Models for Count Data in R. J. Stat. Softw. 27, 1–25. doi:10.18637/jss.v027.i08

Zuur, A.F., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed effects models and extensions in ecology with R, Statistics for Biology and Health. Springer New York, New York, NY. doi:10.1007/978-0-387-87458-6

Appendix A.

Table-A-1. Duration of care before outcome. Short term care 0-7 days, Medium term care 7-28 days, Long term care >28 days.

	n	Min	Max	Average	Numbers in short term	Numbers in medium term care	Numbers in long term care
1996							
Released	8	5	271	94.75	1	1	6
Died in Care	17	0	378	40.82	9	5	3
Euthanized	4	7	136	49.25	0	2	2
1997							
Released	13	0	715	143.85	3	0	10
Died in Care	16	0	2720	203.56	7	3	6
Euthanized	0	NA	NA	NA			
1998							
Released	15	0	1076	189.73	2	3	10
Died in Care	9	0	365	55.44	6	0	3
Euthanized	0	NA	NA	NA			
1999							
Released	19	0	4919	392.26	4	1	14
Died in Care	29	0	243	29.79	12	6	11
Euthanized	4	6	1096	449.75	1	1	2
2000							
Released	21	0	3653	293.19	6	4	11
Died in Care	39	0	1361	68.41	10	11	8
Euthanized	3	4	28	13.67	1	1	1
2001							
Released	33	0	3798	251.94	6	3	24
Died in Care	23	0	1463	118.13	12	3	8
Euthanized	3	0	23	14.33	1	2	0
2002							
Released	24	1	312	80.25	3	2	19
Died in Care	21	0	63	13.76	13	4	4
Euthanized	6	4	731	170.67	1	0	5
2003							
Released	31	0	439	58.35	9	9	13
Died in Care	50	0	573	26.18	29	16	5
Euthanized	4	4	77	44	1	0	3
2004							
Released	27	0	200	55.78	9	2	16
Died in Care	39	0	2922	117.18	22	4	13
Euthanized	3	7	45	28	0	1	2
2005							
Released	30	0	557	90.27	9	2	19
Died in Care	50	0	584	33.22	23	15	12
Euthanized	6	0	1440	246.33	3	2	1
2006							
Released	41	0	538	57.39	15	6	20
Died in Care	83	0	1020	31.70	46	21	16
Euthanized	14	0	56	8.93	8	5	1

Table-A-1 Continued

2007							
Released	60	0	1098	125.63	9	6	45
Died in Care	130	0	292	16.47	75	35	20
Euthanized	32	0	1437	63.75	15	6	11
2008							
Released	50	0	714	119.58	15	9	26
Died in Care	92	0	440	32.11	53	20	19
Euthanized	39	0	48	5.92	31	5	3
2009							
Released	58	1	406	65.95	4	13	41
Died in Care	79	0	233	22.21	41	23	15
Euthanized	91	0	994	21.05	66	17	8
2010							
Released	56	0	191	63.80	9	11	36
Died in Care	47	0	278	19.60	29	12	6
Euthanized	61	0	70	7.20	43	14	4
2011							
Released	151	0	535	74.46	32	10	109
Died in Care	117	0	384	17.18	67	30	20
Euthanized	101	0	371	18.48	63	27	11
2012							
Released	117	1	514	103.20	14	10	93
Died in Care	152	0	203	17.35	91	33	28
Euthanized	61	0	395	44.62	32	8	21
2013							
Released	121	0	593	84.35	15	14	92
Died in Care	156	0	290	14.72	89	42	25
Euthanized	48	0	195	19.69	32	7	9
Total							
Released	875	0	4919	105.3	165	106	604
Died in Care	1139	0	2922	30.89	634	2853	222
Euthanized	480	0	1140	31.51	298	98	84

Table-A-2. Duration in care before outcome for each reported cause of stranding.

Cause of stranding	Died in Care	Euthanized	Released
Boat Strike/Fractures	**49**	**60**	**37**
0-7	29	45	5
7-28	9	4	4
>28	11	11	28
Depredation	**7**	**1**	**7**
0-7	3	1	4
7-28	3	0	1
>28	1	0	2
Disease	**263**	**149**	**49**
0-7	137	94	12
7-28	68	27	2
>28	58	28	35
Dredging	**1**	**0**	**0**
0-7	1	0	0
7-28	0	0	0
>28	0	0	0
Entangled Ghost fishing	**0**	**1**	**0**
0-7	0	1	0
7-28	0	0	0
>28	0	0	0
Entanglement Crabbing	**2**	**2**	**16**
0-7	1	2	3
7-28	1	0	1
>28	0	0	12
Entanglement fishing	**16**	**12**	**30**
0-7	8	10	5
7-28	5	1	7
>28	3	1	18
Entanglement rope	**5**	**2**	**4**
0-7	5	1	0
7-28	0	1	1
>28	0	0	3
Ingestion of foreign material	**35**	**18**	**1**
0-7	15	10	1
7-28	16	5	0
>28	4	3	0
Netting	**0**	**1**	**3**
0-7	0	1	2
7-28	0	0	0
>28	0	0	1
Other Anthropogenic	**9**	**2**	**1**
0-7	1	2	0
7-28	3	0	0
>28	5	0	1
Shark Control Program	**4**	**1**	**6**
0-7	2	0	2
7-28	2	0	0
>28	0	1	4

Table-A-2 Continued

Unknown	**744**	**231**	**361**
0-7	430	131	93
7-28	174	60	50
>28	140	40	218
Unknown Natural	**4**	**0**	**2**
0-7	2	0	2
7-28	2	0	0
>28	0	0	0
Buoyancy Disorder	**0**	**0**	**358**
0-7	0	0	36
7-28	0	0	40
>28			282
Grand Total	**1139**	**480**	**875**